APPENDICES

A.1.0 EXTERNALITY ADDERS (UNIT DAMAGE COSTS) USED IN THE ANALYSIS

INTRODUCTION

There are a number of studies that have looked at the external costs associated with different air pollutants (in particular). We have not made a complete investigation of these. For the most part, so as to avoid replicaton with studies already carried out for the European Commission, we have used figures which have been used in earlier studies (COWI 2000; Brown et al 2000). The reader is referred to these also for further explanation of the choice of externality adders used.

In general, we have taken the view that the use of ranges is warranted. This is an area of economics where certain assumptions are controversial, and debates are unresolved. This, allied to the fact that the science underpinning 'what is being valued' is frequently characterised by uncertainty, suggests that the degree to which unit damage cost estimates can be said to be 'known with certainty' in any specific context and for any given pollutant is extremely limited.

These problems, which are significant in their own right, are magnified when, as in this study, one is seeking estimates that can be considered as 'transferable' across different locations and processes. For example, it is obviously not sound to expect the unit damage costs of emissions from 50m high stacks (e.g. incinerators) to be the same as those emitted at a lower level (e.g. landfill flares). On the other hand, the population densities around these facilities might be expected to be higher and lower, respectively, such that where pollutants with more local impacts are concerned, the two effects act to counterbalance each other.

The watch word is caution. All of the results derived must be treated as approximations, and given the fact that valuation of all emissions is not possible, the results reported in later Appendices are also partial.

REVIEW OF ESTIMATES, AND ADDERS USED

Particulate Matter (PM_{10})

We have looked at several European studies (see Table 1). For comparison, RPA and Metroeconomica (2000) site US-based studies by Rowe et al (1995) and Thayer et al (1994) which give values of 20,534 ECU per tonne and 46,825 ECU per tonne, respectively.

Note that some estimates are for all particulates or Total Suspended Particulate matter (TSP), whilst some are for PM_{10} specifically. The range suggested by the review below is, approximately, from €8,000 per tonne to €70,000 per tonne. Not all these studies are strictly comparable – they tackle pollutants arising in different contexts. The variation is therefore very significant. The values given in studies for transport in urban areas tend to be especially high (see the ECMT (1998) and Pearce and Crowards (1995) figures).

The ExternE programme gave values for all different EU countries (see Table 2). The ranges quoted are based on mortality estimates valued using the Years of Life Lost approach which was the one recommended in the study. This remains a matter of considerable debate (though it was the recommended approach under ExternE). Since the mortality component of the total damage cost tends to be significant, and since the Value of Statistical Life approach increases this component three-to-four fold (relative to the Years of Life Lost approach), these may be relatively low estimates.

Note that the study itself points out that the damages for incinerators, the principal source of PM_{10} in our study, are likely to be at the high end because of the significance of large populations in the valuation approach (the greater the population exposed, the greater the damage). Incinerators tend to be located in larger cities.

Note also that some studies have suggested that in the case of incinerators, particulate matter tends to be of the sub-2.5 micron type (which is more likely to be damaging to human health because of its effect on respiratory functions). Dockery and Pope (1994) suggested that a factor of 1.67 might be appropriate. This makes it possible that one should be using substantially higher unit damage costs than those quoted here.

Table 1 Estimates of Damages from Recent European Studies

Study	Study Area	Pollutant	Damage		
			Low	**Central**	**High**
Krewitt *et al* (1997) (ECU per tonne)	UK/Germany[1]	Particulates	22046		60439
CSERGE (1993) (ECU per tonne)	UK	Particulates		12240	
AEA (1997) (ECU per tonne)	UK incinerator (50m stack)	PM_{10}			
AEA (1997) (ECU per tonne)	UK incinerator (90m stack)	PM_{10}			
AEA (1997) (ECU per tonne)	UK incinerator (100m stack)	PM_{10}			
Pearce and Crowards (1995) (£ per tonne)	UK	PM_{10}	23288		57748
Beukering et al (1998)	EU	PM_{10}		20468	
ECMT (1998) (ECU/tonne)	UK (rural transport)	PM_{10}		0	
ECMT (1998) (ECU/tonne)	UK (urban transport)	PM_{10}		70000	
Powell et al (1996) (£/tonne)	UK	PM_{10}		8980	
Coopers and Lybrand et al (1997) (ECU/tonne)	UK	TSP (transport)		7522	
Coopers and Lybrand et al (1997) (ECU/tonne)	UK	TSP (electricity generation)		12149	

Table 2 Damages from Particulates ExternE National Implementation Studies

Country	Particulates
Austria	16,800
Belgium	24,536-24,537
Denmark	3,390-6,666
Finland	1,340-2,611
France	6,100-57,000
Germany	19,500-23,415
Greece	2,014-8,278
Ireland	2,800-5,415
Italy	5,700-20,700
Lux	As Netherlands
The Netherlands	15,006-16,830
Portugal	5,565-6,955
Spain	4,418-20,250
Sweden	2,372-3,840
United Kingdom	8,000-22,917

The High and Low values used in the study are those represented by the ranges in Table 2 above. For East European countries, we have simply taken average values for the high and low figures.

Sulphur Dioxide

See Table 3 for studies reviewed. Note that not all studies include all effects. Dorland (1997), in work in the Netherlands, suggested a value for acute mortality impacts of 629 ECU per tonne from direct effects only. The greater impact came from indirect effects associated with sulphate aerosols, the valuation figure being 6953 ECU per tonne. Dorland estimated the effects of stack height upon the valuation and found that the indirect effects, which dominate, were not affected by stack height whilst the direct effects did vary with stack height. These could be 1.5 times greater for ground level emissions than for those from a 175 metre high stack.

Table 3 Estimates of Damages from Recent European Studies

Study	Study Area	Pollutant	Damage		
			Low	Central	High
AEA (1997) (ECU per tonne)	Birmingham, UK (50m incinerator stack)	SO_2		20131[a]	
AEA (1997) (ECU per tonne)	Birmingham, UK (90m incinerator stack)	SO_2		18715[a]	
AEA (1997) (ECU per tonne)	Birmingham, UK (100m incinerator stack)	SO_2		18243[a]	
CIEMAT 1998 (ECU/tonne)	UK	SO_2	6027		10025
Powell et al (1996) (£/tonne)	UK	SO_2		2584	
Coopers and Lybrand et al (1997) (ECU/tonne)	UK	SO_2		4339[b]	
Davidson and Wit (1998) (£/tonne)		SO_2	2000		4000

Notes: [a] Includes acute health, chronic health and materials impacts.

[b] Includes impacts on health, buildings, crops and forests.

The values above can be compared with Member State specific ExternE values below (Table 4). These show much lower values for countries where much of the pollution effectively falls on the sea. This is not to say that there are no consequences following from such pollution, merely that such consequences are not included in the ExternE analysis (or for that matter, most others). Note also, however, that the values also exhibit enormous variation within countries. This suggests that the impacts of local population density, stack height and so forth are quite important in determining ‘real’ impacts (as are local topographical and meteorological factors, less easily captured by the modelling exercise).

Table 4 Damages from Sulphur Dioxide, ExternE National Implementation Studies

Country	Sulphur Dioxide
Austria	9,000
Belgium	11,388-12,141
Denmark	2,990-4,216
Finland	1,027-1,486
France	7,500-15,300
Germany	1,800-13,688
Greece	1,978-7,832
Ireland	2,800-5,300
Italy	5,700-12,000
The Netherlands	6,205-7,581
Portugal	4,960-5,424
Spain	4,219-9,583
Sweden	2,357-2,810
United Kingdom	6,027-10,025

The High and Low values used in the study are those represented by the ranges in Table 4 above. For East European countries, we have simply taken average values for the high and low figures.

Oxides of Nitrogen (NO_x)

See Table 5 and Table 6 for studies reviewed. The values used in this study are, at the low end, €6,500 and at the high end, €40,000.

Table 5 Estimates of Damages from Recent European Studies

Study	Study Area	Pollutant	Damage		
			Low	Central	High
Krewitt *et al* (1997) (ECU/tonne)	UK/Germany	NO_X	17864		47003
CSERGE (1993) (ECU/tonne)	UK	NO_X		1005 [a]	
AEA (1997) (ECU per tonne)	Birmingham, UK (50m incinerator stack)	NO_X		34739 [a]	
AEA (1997) (ECU per tonne)	Birmingham, UK (90m incinerator stack)	NO_X		34267 [a]	
AEA (1997) (ECU per tonne)	Birmingham, UK (100m incinerator stack)	NO_X		34149 [a]	
ECMT (1998) (ECU/tonne)	UK (rural transport)	NO_X		4000	
ECMT (1998) (ECU/tonne)	UK (urban transport)	NO_X		8000	
CIEMAT 1998 (ECU/tonne)	UK	NO_X	5736		9612
Powell et al (1996) (£/tonne)	UK	NO_X		1270	
Coopers and Lybrand et al (1997) (ECU/tonne)	UK	NO_X		3076 [b]	

[a] Includes acute health, chronic health and materials impacts.

[b] Includes impacts on health, buildings, crops and forests.

Table 6 Damages from NOx, ExternE National Implementation Studies

Country	NOx
Austria	9,000-16,800
Belgium	11,536-12,296
Denmark	3,280-4,728
Finland	852-1,388
France	10,800-18,000
Germany	10,945-15,100
Greece	1,240-7,798
Ireland	2,750-3,000
Italy	4,600-13,567
The Netherlands	5,480-6,085
Norway	Na
Portugal	5,975-6,562
Spain	4,651-12,056
Sweden	1,957-2,340
United Kingdom	5,736-9,612

Note: Damages are due to nitrate only, no account taken of damages from ozone associated with NOx.

The High and Low values used in the study are not quite the same as those represented by the ranges in Table 6 above. What we have done is we have increased the unit damage costs (both the high and the low) by €1,500 per tonne to reflect ozone related damages associated with NO_x. (An AEA (1997) report gives a value of 2530 ECU/tonne of ozone. The CIEMAT (1998) report, acknowledging the complexity of the reactions involved, gives a value for the EU of 1500 ECU / tonne NO_x.) Then, for East European countries, we have simply taken average values for the high and low figures. Table 6

Volatile Organic Compounds

Estimates for damage costs from volatile organic carbons do not always obviously include estimates for creation of tropospheric ozone. For Volatile Organic Carbon compounds, ECMT (1998) use a figure of 4000 ECU/tonne in rural areas and 8,000 ECU per tonne in urban areas.

We have used the figures used in the COWI study, which are, in turn, based upon a review of literature. The range used was from €757/tonne to €1,500 per tonne of VOCs. It is difficult to say how 'good' tese estimates are, not least because VOCs themselves are a heterogeneous group of compounds, varying significantly in their potential to cause harm.

Greenhouse Gases (Carbon Dioxide, Methane and Nitrous Oxide)

Evidently, placing values on greenhouse gas emissions presents particular problems. Theoretically, one needs to know how climate will change because of anthropogenic emission of gases (relative to the counterfactual). The uncertainty surrounding climatic projections and the dynamic path by which climate changes, specifically, the frequency and severity of extreme events, makes easy quantification a rather distant prospect.

Carbon Dioxide

Marginal Social Costs for CO_2 emissions from a number of studies are given in Fankhauser and Tol (1995). Note that these vary over time so that typically, the shadow price of a tonne of CO_2 rises over time. Where ranges were given, they were given for 90% confidence intervals. Examples of these are:

- from Nordhaus (1991) $0.3-$65.9;
- from Cline (1992) $5.8-$124; and
- from Fankhauser (1994) $6.3–$45.2.

All these are valued in $1990 and are per tonne of carbon (so for values for CO_2, one has to multiply by the relative molecular weights, that is (12/44). Other studies include ECMT (1998) which, in the spirit of precautionary approach, used 50,000 ECU / tonne CO_2. Davidson and Wit (1997) (cited in ECOTEC 1999) estimate damage costs at £30 / tonne CO_2. Ecobalance and Dames and Moore (1999) used £3-109 in their recent report for the UK's DTI. The ExtenE programme of research has led to a number of estimates. We have sought to nclude some functionality within our modelling by enabling global warming damages – which occur over extended periods of time – to be made sensitive to the discount rate chosen.

Methane

The two extreme values that we have made use of effectively come from Fankhauser (1995) and Davidson and Wit (1997). Fankhauser's range for a 90% confidence interval is £36.6-136.4 /tonne CH_4 . This was the range used in work done for us by CSERGE in ECOTEC (1999). The same study mentioned the work by Davidson and Wit (1997).

Nitrous Oxide

Estimates can be found from earlier work carried out for us by CSERGE (in ECOTEC 1999). The values from Fankhauser (1995) cover a 90% confidence range with a low value of £614.30, the high value, £5,534.78 per tonne of N in N_2O.

Values Used in the Study

Based upon ExternE work, we have chosen the unit damage costs shown in Table 7. These have been taken from ranges estimated in the two models assessed in the ExternE programme.

Table 7:Unit Damage Costs for Greenhouse Gases

Discount Rate	CO2	€/tonne	CH4	€/tonne	N2O	€/tonne
1%	Low	51.4	Low	466.9	Low	19048.7
	High	53.7	High	618.6	High	29133.3
3%	Low	24.5	Low	450.6	Low	8239.6
	High	25.7	High	489.2	High	14161.8
5%	Low	6.7	Low	163.3	Low	2117.2
	High	9.6	High	231.3	High	3016.6

Note that in much of the global warming modelling, it is not discount rates per se that are being used to understand the basis for variation in damages, but the pure rate of time preference. The to are not the same.

Carbon Monoxide

Damage costs for carbon monoxide are given in Fankhauser (1994). The central estimate as given in Powell et al (1996) is 0.6p/kg, or £6 per tonne. We have used the values used by COWI (2000) of €2-€9 per tonne as high and low values.

Heavy Metals and Dioxins

A fairly comprehensive treatment of benefits assessment associated with heavy metals from incineration plants under different assumptions is given in AEA (1997). The reader is directed there for details of the derivations and the discussions surrounding specific pollutants. The greatest variation is witnessed in the case of dioxins. Here, the assumption concerning the absence or otherwise of thresholds has enormous influence on the results. The values we have used are those employed by COWI (2000) (all values are €/tonne):

- For dioxins, a low of € 2,339,717,000, a high of € 17,630,080,000;
- For cadmium, a low of €20,000 and a high of €95,000
- For arsenic, a low of €162,000 and a high of €1,168,000;
- For mercury, AEA use a value of €0 and COWI give no value;
- For chromium, a low of €133,000 and a high of €958,000;
- For nickel, a low of 3,000 and a high of €20,000.

For chromium and nickel, and most obviously, mercury, the values seem quite low. For lead, we have used a range from EFTEC (1996) of €4890-€14670 per tonne.

Others

We do not have data on emissions from any of the treatment routes for CFCs, and we do not feel that the valuation work available allows for an easy quantification of impacts from water pollution. These are omitted from the valuation work undertaken.

A discusson concerning leachate can be found in the Appendix 2. Appendices 2 and 3 also deal with disamenity from landfills and incinerators, whilst Appendix 3 discusses the avoided extrnalities associated with aggregates replacement.

Other external costs and benefits are treated in relevant Appendices.

A.2.0 EXTERNAL COSTS OF LANDFILL

INTRODUCTION

The model for the external costs of landfill builds on the work undertaken by Broome et al (2000) for Waste Watch and Friends of the Earth (the Waste Watch study), and that undertaken for the Commission by COWI (2000) (the COWI study), but there are marked improvements upon these analyses. In both studies, a review of landfill externalities was undertaken. The reader is referred to these reports for further details.

In the first of these studies, the key variables driving the external cost analysis were:

- Methane generated by the waste landfilled (and the model was set up to handle different waste compositions);
- Oxidation rate of methane through the cap;
- Landfill gas collection efficiency;
- Where energy is recovered from the landfill gas;
    - the efficiency of the energy recovery process; and
    - the assumption made concerning any displacement of burdens associated with fuels assumed to be 'displaced' by the energy from the waste.

In the second study, the key externalities were:

- Disamenity;
- Global warming emissions especially methane; and
- Where energy is recovered, avoided pollution.

The largest externality in the study is that from disamenity which, as the report notes, is based upon a meta-analysis of US studies relating to disamenity. This is highly sensitive to the assumptions used concerning the average household density in the vicinity of the landfill and average (without-landfill) real estate prices. As regards the treatment of other externalities such as leachate, the COWI report made use of two reports to attribute leachate externalities which were assumed to be between zero and €1.01 for a well-operating landfill, and between €1.01 and €2.03 per tonne of MSW for older sites without liners (CSERGE *et al* 1993; Miranda & Hale 1997).

The Waste Watch study did not look at leachate but recognised that this may be a significant problem, if not today, then in the future.[1] Liabilities for clean-up could be significant in the event of hazardous materials leaching from landfills. Other omissions identified in the Waste Watch study were:

- All the relatively fixed externalities, such as the impacts associated with landfill construction and engineering, any changes in non-use values of specific sites, and possibly, any non-market benefits from recreational uses post-closure (though these might have to be considered against counterfactual land-uses);

- All impacts associated with the use of on-site vehicles;

- Emissions of gases other than CO_2 and CH_4 (ozone depleting chemicals, such as CFCs, are believed to arise from landfills); and

- A number of other impacts whose status is 'unproven' as yet, for example, the possible problems in respect of birth defects that have been mentioned in the context of landfilling

Also noted was the fact that the possibility remains for heavy metals (from, for example, fluorescent tubes) to enter water courses through breaching landfill liners in the future. This is possibly one example of the 'low probability, high consequence risks' which social theorists have recently sought to come to terms with. Another example would be explosions from gas pockets in landfill sites. All of the above (apart from the possible benefits from non-market recreation and amenity post-closure – likely to be heavily discounted) are negative externalities. As such, the net externalities quoted by

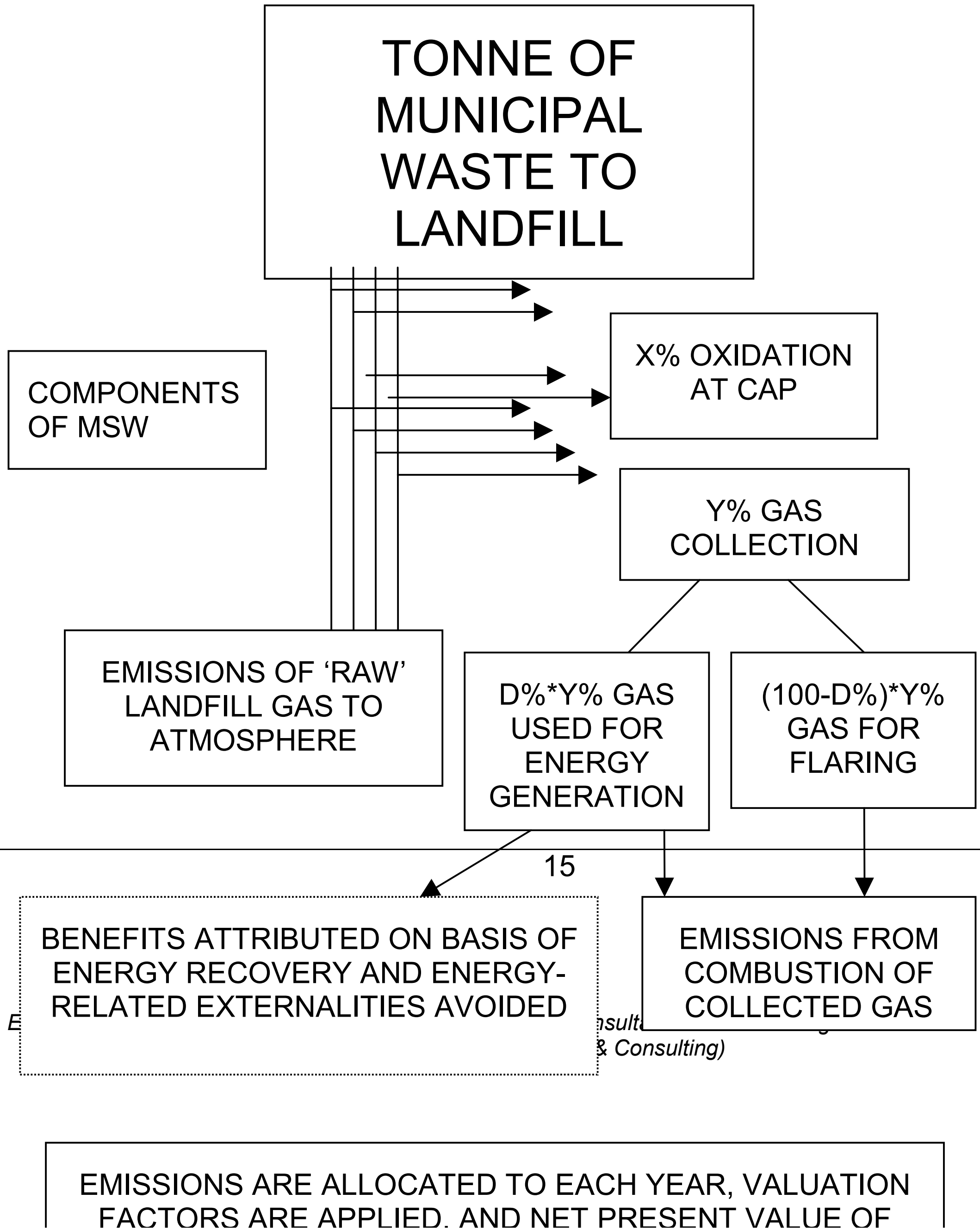
[1] The study by Bez et al (1998) lists 23 different emissions to water as well as emissions to air of VOCs, hydrogen sulphide, sulphur dioxide, dust, carbon monoxide and NO_x. These were from landfilling contaminated bottle fractions only.

both the Waste Watch study and the COWI study probably both paint a more positive picture of the true situation regarding landfill than is warranted. Several aspects and pollutants are not subject to valuation and these tend to be factors which are overwhelmingly negative in the general implications. In addition, the COWI report is at least suggestive of the fact that disamenity may be a significant component of the external costs associated with landfill. This is of some significance since life-cycle studies, which are frequently used to assess the environmental impacts of waste treatment options, have absolutely nothing to say about disamenity effects.

This Analysis

Figure 1 below shows the approach undertaken in the modelling. Note the way in which gaseous emissions are treated. The unit damage costs for greenhouse gases are also adjusted to reflect the choice of discount rate. Quantitative data for emissions are given in the following sub-sections.

Figure 1 Modelling of Landfill Externalities in the Study



Air Emissions

Emissions to air from landfill are assumed to come from landfill gas (LFG). Fugitive emissions of 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 these tend to be small relative to the total emissions for modern large-scale landfills (see Gregory et al 2000). On the other hand, they do occur, by definition, at an earlier time, so if one was discounting emissions heavily, they may in fact be assume greater importance from an economic perspective.

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 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 equations used in this analysis are, therefore:

$$Emissions = \sum DOC*FCD*e^{-kt}$$

Where DOC is the degradable organic carbon fraction, FCD is the fraction of organic carbon dissimilable, 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 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. However, the attempt here is to ensure that the significance of the time dimension is captured in the analysis, so as a first order approximation, the model should perform acceptably within the bounds of existing knowledge.

Table 8). These are based on work undertaken by the IPCC and by AEA Technology for the European Commission (IPPC u.d.; AEA 2001). They have been validated to some extent through assessing the implied methane emissions from the materials and cross-checking against work undertaken in the United States and by the UK Environment Agency (Barlaz 1997; Gregory and Revans 2000).

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. However, the attempt here is to ensure that the significance of the time dimension is captured in the analysis, so as a first order approximation, the model should perform acceptably within the bounds of existing knowledge.

Table 8 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	0.2	12%	75%
Garden waste	0.1	22%	50%
Average putrescibles		19%	64%
Fines	0.06	9%	60%
Wood	0.03	28%	35%

Source: Figures were based upon a review of IPPC (u.d.) and AEA (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.

In the study, we have assumed that landfill gas is always (for each component and in each year) 50% of the total. 'Raw' methane and carbon dioxide emissions are then converted to emissions to the atmosphere using the following (variable) parameters:[2]

- The overall efficiency of gas collection;
- The percentage of uncaptured methane which is oxidised (to carbon dioxide) in leachate and at the cap;
- 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, it is argued that this is an improvement upon earlier attempts to model the external costs of landfill emissions. The principal advantages of this approach are:

- That the effect of different discount rates (implying different unit damage costs for greenhouse gases, both in absolute and relative terms) upon the externality calculus can be assessed; and

[2] 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.

- That the effect of changing the composition of materials landfilled can also be assessed.

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 below. The way this is done is to normalise the concentrations (by weight) so that:

- With the emissions of CO_2 re-based to (88/16) (the relative weight of carbon dioxide emitted to methane captured, under the assumption that half the gas is carbon dioxide), the weights of trace gases associated with flaring / energy recovery are derived by multiplying by the quantity of methane captured for flaring / energy generation; and
- With the CH_4 emissions set to 1, in the uncollected landfill gas case, the relative emissions of other gases in uncollected landfill gas are calculated by multiplying the relative proportions of the gases in uncollected landfill gas by the quantity of uncollected methane.

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. This makes the data in Table 9 somewhat strange since it suggests that more than 50% (actually, 55%) of the uncollected landfill gas is methane.

Table 9 Air Emissions From Landfilling, Including Landfill Gas And Flare/Engine Exhaust (All Amounts Are Given In Mg/Nm3 Of LFG).

mg/N m3	Flare/engine exhaust	Landfill gas
CH_4	-	392,860
CO_2	1,964,290	883,930
CO	800	13
H_2S	0.033	200
HCl	12	65
HF	0.021	13
HC	60	2000
Chlorinated HC	10	35
Dioxins	0.0000008	-
PM_{10}	4.3	-
NO_x	100	-
SO_x	25	-
Cd	0.0000094	0.0056
Cr	0.0000011	0.00066

Pb	0.0000085	0.0051
Hg	0.000000069	0.000041
Zn	0.00013	0.072

Source: From White et al (1995).

Emissions of Leachate to Soil and Water

In the externality assessment, we will use the assumptions made in the COWI report, though in practice, these have (as that study notes) significant limitations. COWI indicate the considerable variation in the composition of leachate. For the sake of completeness, we show these physical compositions below in Table 10 and Table 11.

Table 10 Typical Ranges In Observed Leachate Composition From MSWLandfills (all values in mg/l leachate, except pH and conductivity).

	Low	High
Conductivity (µS/cm)	2,500	35,000
SS	200	60,000
Organic substances:		
TOC	30	29,000
BI5	20	57,000
COD	140	152,000
Organic N	14	2,500
Inorganic macrocomponents:		
Total P	0.1	23
Chloride	150	4,500
Sulphate	8	7,750
Hydrogen carbonate	610	7,320
Sodium	70	7,700
Potassium	50	3,700
NH4-N	50	2,200
Calcium	10	7,200
Magnesium	30	15,000
Iron	3	5,500
Manganese	0.03	1,400
Silicate	4	70
Heavy metals:		
As	0.01	1
Cd	0.0001	0.4
Cr	0.02	1.5
Co	0.005	1.5
Cu	0.005	10
Pb	0.001	5
Hg	0.00005	0.16
Ni	0.015	13
Zn	0.03	1,000

Source: Ranges based on observed data from international literature for MSW landfills (from Christensen et al, 1994, in COWI 2000).

Table 11 Typical Ranges For The Most Frequently Observed Specific Organic Compounds In Leachate From MSW Landfills (all values in mg/l leachate).

mg/l	Low	High
Aromatic Hydrocarbons:		
Benzene	1	1,630
Toluene	1	12,300
Xylene	4	3,500
Ethylbenzene	1	1,280
Trimethylbenzene	4	250
Naphthalene	0.1	260
Diethylphthalate	10	660
Di-n-butylphthalate	5	15
Butyl-benzyl-phthalate	5.1	8
Chlorinated Hydrocarbons:		
Chlorobenzene	0.1	110
1,2-Dichlorobenzene	0.1	32
1,4-Dichlorobenzene	0.1	16
1,1-Dichloroethane	0.6	46
1,2-Dichloroethane	<6	
1,1,1-Trichloroethane	0.1	3,810
trans-1,2-Dichloroethylene	1.6	88
cis-1,2-Dichloroethylene	1.4	470
Trichloroethylene	0.7	750
Tetrachloroethylene	0.1	250
Methyl chloride	1	64
Chloroform	1	70
Carbon tetrachloride	4	9
Phenols:		
Phenol	1	1,200
Ethyl phenols	<300	
Creosols	1	2,100
Pesticides:		
MCCP	2	90
2,4-D	1	5
Other:		
Acetone	6	4,400
Tetrahydrofuran	9	430
Methylethylketone	110	6,600
Tri-n-butylphosphate	1.2	360
Triethylphosphate	15	
Camphor	Identified	
Fenchone	20	34

Notes: 2,4-D = 2,4-Dichlorophenoxyacetic acid; MCCP = 2-(2-methyl-4-chlorophenoxy) propionic acid.

Source: Above ranges based on observations at 17 sites (from Christensen et al, 1994 in COWI 2000).

However, it important to stress that they have no bearing on the external cost analysis. This is somewhat worrying, but in our view (and to some extent, it would seem also, that of COWI), there is no satisfactory approach for dealing with leachate externalities. It is, however, worth pointing out that the leaching of toxic elements could potentially generate significant problems in the future. Arguably, we simply do not know whether this will indeed be the case.

Emission 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 Table 12 and Table 13

Table 12 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
Organic substances:				
TOC	-	2,177	5	4,350
BOD	-	4,277	3	8,550
COD	-	11,411	21	22,800
Organic N	-	189	2	375
Inorganic macrocomponents:				
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
Heavy metals:				
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)

Table 13 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
Aromatic Hydrocarbons:				
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
Chloromatic Hydrocarbons:				
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
Phenols:				
Phenol	-	90	0.2	180
Ethyl phenols	-	45	45	45
Creosols	-	158	0.2	315
Pesticides:				
MCCP	-	7	0.3	14
2,4-D	-	0.5	0.2	1
Other:				
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

Notes: Best estimates are calculated as the arithmetic mean of the high and low estimates

Source: COWI (2000)

Disamenity

Disamenity provides another good example of the difficulties of undertaking this type of external cost analysis. Assumptions concerning avoided energy use have tended to adopt a marginalist analysis. If one was to adopt a similar marginalist approach to disamenity, one would probably have to conclude that the disamenity associate with landfilling a marginal tonne of waste is approximately zero. Yet most studies adopt an 'average' approach, implicitly acknowledging that much of the disamenity probably arises from the existence of the landfill rather than the magnitude of deposits per se (indeed, for a landfill of given size, the speed at which the site is filled clearly affects the period over which disamenity is experienced, and depending upon the restoration plans, the period after filling is likely to be one in which the landfill guarantees a lack of development that might otherwise occur).

The disamenity from landfills was not estimated in the Waste Watch study, but the COWI study used the meta-analysis of hednic pricing studies carried out by Brisson and Pearce (1995). This suggests that house prices within a given radius are affected by the presence of the landfill site, this falling off to zero in line with the equation:

$$\Delta \text{ (House Price, as \%)} = -(12.8 - 3.76 * R)$$

Where R is the distance from the landfill in miles (one mile = 1.6 km). Converting this to kilometres, the equation can be written:

$$\Delta \text{ (House Price, as \%)} = -(12.8 - 2.35 * r)$$

Where r is the distance from the landfill in kilometres.

Hence, the condition for the effect to fall to zero is:

$$R = (12.8 / 2.35) = 5.26 \text{ kilometres.}$$

The total change in house values, in the area, can therefore be approximated by integrating over what are actually discrete annuli (rings) around the landfilled site:[3]

[3] The integration provides an approximation since in fact the distribution of houses in the area affected by the landfill is more properly represented by a discrete variable as opposed to a continuous one. This means, in practice, that the lower is the population density, the greater will be any errors associated with reducing the discrete distribution of households by a continuous function.

$$-\int_{0 \text{ to } 5.26} P(12.8-2.35r)*\rho*2\pi r.\delta r$$

where P is the ‘without landfill’ house price and ρ is the housing density in number of households per square kilometre. Both of these may, of course, exhibit some functional relationship to R, but assuming that they are constant over the affected area, the result becomes:

$$-P*\rho*2\pi*\int_{0 \text{ to } 5.26} (12.8-2.35r)*r.\delta r$$

[4]

$$= -P*\rho*2\pi*[6.4r^2-0.78r^3]$$

$$= -P*\rho*2\pi* r^2 [6.4-0.78r]$$

It becomes clear that the change in overall housing values is dependent upon the average housing price and the density of households around the site.[5]

In our view, the calculations in the COWI report are incorrect. Either:

- The annualised values, assumed to be 8% of the total, are too high and it may be that the calculations have been performed using ‘kilometres’ rather than miles; or

[4] Note, it is not clear whether the integration should start from a ‘zero radius’. This depends strictly upon whether ‘r’ is the distance from the landfill perimeter or from the landfill’s centre. Arguably, if it is the latter, one should take the integral not from ‘r=0’, but from a value of r which represents the radius of the landfill itself.

[5] Note that the effects on more general real estate values may be different. The type of real estate around landfills might not necessarily be ‘housing’, but commercial real estate. It is not clear how this should be accounted for in such an analysis as this (see below concerning the effects of incinerators).

- The figures quoted are total values and are too low by a factor 2π (this seems less likely given that the per tonne figures are higher than calculated by us – see Table 14 below).

Given the significance of disamenity in the anaysis, this is potentially highly significant. The net externality is approximately equivalent to the total externality in that report. Adjusting the externality for the incorrect calculation would give a negative externality associated with disamenity of the order €5 per tonne (under the best estimate). This is half the original value given.

Table 14 Disamenity Figures For Landfills

Houses per square mile	Average house price, €	Total disamenity value (COWI, €)	Total disamenity value (adjusted calculation, €)	Annual disamenity value (adjusted calculation, €)
80	100,000	2,001,026	12,424,842.37	993,987.4
80	50,000	1,000,513	6,212,421.19	496,993.7
80	200,000	4,002,052	24,849,684.74	198,7975
40	50,000	500,257	3,106,210.59	248,496.8
160	200,000	8,004,104	49,699,369.49	3,975,950

Source: Author's calculations based on COWI (2000)

Note the best estimate is based upon a housing density of 80 per square mile (or 31 per square kilometre) and an average house price of €50,000. The population density around landfill sites is likely to vary considerably across Member States, whilst the average housing price (remembering that it should be the 'without-landfill' price being quoted) will also vary considerably across countries. These points suggest that the best estimate needs considerable revision to account for cross-country variation in housing prices (which may be influenced by institutions in the Member States concerned) and for variations in population density in the proximity of landfill sites. In this study, without better knowledge of these, we use the figure €5 per tonne as the figure around a landfill of 80 households per square mile. We assume, for each Member State, for low and high estimates of disamenity, an average population density around landfills of one- and two-thirds the country average, respectively (see Table 15), and multiply this by (€5/80). This gives the figures in Table 16. Clearly, these show no variation according to average house price and size of landfill. There is, however, substantial variation across Member States on the basis of the variation in population density.

Table 15 Population Density Assumptions And Landfill Disamenity Externalities By Country

	Inhabitants per sq.km			Inhabitants per sq. mile			Inhabitants per hhld	Households per sq. km			Households per sq mile		
	1998	1991		1998	1991			1998	1991		1998	1991	
		Max	*Min*		*Max*	*Min*			*Max*	*Min*		*Max*	*Min*
EU-15	117			300			2.62	45	0	0	114	0	0
A	96			246			2.5	38	0	0	98	0	0
B	334	5891	53	855	15081	136	2.5	134	2356	21	342	6032	54
DK	123			315			2.2	56	0	0	143	0	0
FIN	15			38			2.2	7	0	0	17	0	0
FR	108	904	2	276	2314	5	2.6	42	348	1	106	890	2
GER	230	3887	80	589	9951	205	2.3	100	1690	35	256	4326	89
GRE	80	923	31	205	2363	79	3.1	26	298	10	66	762	26
IRL	54			138			3.3	16	0	0	42	0	0
IT	191	416	36	489	1065	92	2.8	68	149	13	175	380	33
LUX	164			420			2.6	63	0	0	161	0	0
NL	382	953	105	978	2440	269	2.4	159	397	44	407	1017	112
P	108	319	20	276	817	51	3.1	35	103	6	89	263	17
SP	78	4102	21	200	10501	54	3.3	24	1243	6	61	3182	16
SWE	22			56			2.1	10	0	0	27	0	0
UK	244	4376		625	11203	0	2.5	98	1750	0	250	4481	0

Sources: Inhabitants per household from European Environment Agency (1995); population density from Eurostat (2000).

Table 16 Low And High Estimates Of Landfill Disamenity In Member States

	Avge. Hhld density	Low (€ per tonne)	High (€ per tonne)
A	98	2.04	4.08
B	342	7.13	14.25
DK	143	2.98	5.96
FIN	17	0.35	0.71
FR	106	2.21	4.42
GER	256	5.33	10.67
GRE	66	1.38	2.75
IRL	42	0.88	1.75
IT	175	3.65	7.29
LUX	161	3.35	6.71
NL	407	8.48	16.96
P	89	1.85	3.71
SP	61	1.27	2.54
SWE	27	0.56	1.13
UK	250	5.21	10.42

Avoided Burdens due to Energy Production

The approach to calculating externalities associated with avoided energy use is, as discussed above, undertaken using country-specific avoided external costs where it being assumed that there is a net displacement effect. The electricity generated is calculated from the partitioning of total gas into that which is uncollected, flared and used for energy recovery. Of the recovered fraction, it is assumed that each tonne of methane gives up 50.4 GJ of energy (or 14,000 kWh per tonne, or 36 MJ/m3 methane). Of this collected fraction, the fraction generated as electricity depends upon the efficiency of transformation of this energy, which is a parameter which can be entered into this model. Typical efficiencies appear to be of the order 30%. This is the default figure used.

RESULTS

We present below, for the Italian case:

- The 'default' landfill (10% oxidation at the cap, 60% landfill gas collection efficiency, 50% of collected gas used for energy generation and 30% engine efficiency) at differing rates of discount (1%, 3% and 5%) (Table 17, Table 18 and Table 19);
- At the 1% discount rate, a 'worst case' landfill and a 'best case' landfill to show the effects of changing the parameters used in the default case (Table 20 and Table 21).

The insights from these illustrations are as follows:

- As the discount rate chosen increases, so the global warming externalities fall. Equally, so do the benefits associated with energy recovery. The net effect is that the external costs of landfilling fall as the discount rate increases. This is what one would expect;
- As the proportion of gas collected increases, and as the fraction of uncollected gas which is oxidised increases, so the balance of global warming externalities shifts away from methane and towards carbon dioxide. This reduces the overall damages associated with greenhouse gases, again as expected; and
- As the proportion og gas used to recover energy increases, and as engine efficiency increases also, so the net benefit from energy generation increases;
- The externalities associated with disamenity and leachate are invariant in this analysis whilst the impact of 'other air pollutants' remains very small independent of the assumptions made (though it does vary).

Hence, the key factors influencing the analysis are the discount rates applied and the parameters used to characterise landfill performance.

Table 17 Externalities of High Quality Landfill in Italy, 1% Discount Rate

Country	ITALY		
Oxidation at Cap			**10.00%**
Landfill Gas Collection Efficiency			**60.00%**
Used to Generate Energy			**50.00%**
Engine Efficiency			**30.00%**
Discount Rate			**1.00%**
		Low	**High**
Global Warming Effects			
	Methane	-5.96	-7.89
	Carbon Dioxide	-8.21	-8.58
Other Air Pollution		-0.04	-0.10
Leachate		-1.00	-2.00
Net External Cost		***-15.21***	***-18.58***
Avoided Burdens (Energy Production)		2.58	3.18
Disamenity		-3.65	-7.29
Net Externality (excl. disamenity, incl. displaced burdens from energy generation)		***-12.63***	***-15.40***
Net Externality (excl. disamenity, excl. displaced burdens from energy generation)		***-15.21***	***-18.58***
Net Externality (incl. disamenity, incl. displaced burdens from energy generation)		***-16.28***	***-22.69***
Net Externality (incl. disamenity, excl. displaced burdens from energy generation)		***-18.86***	***-25.87***

Table 18: Externalities of High Quality Landfill in Italy, 3% Discount Rate

Country	ITALY		
Oxidation at Cap			**10.00%**
Landfill Gas Collection Efficiency			**60.00%**
Used to Generate Energy			**50.00%**
Engine Efficiency			**30.00%**
Discount Rate			**3.00%**
		Low	**High**
Global Warming Effects			
	Methane	-4.49	-4.87
	Carbon Dioxide	-3.05	-3.21
Other Air Pollution		-0.03	-0.08
Leachate		-1.00	-2.00
Net External Cost		***-8.58***	***-10.17***
Avoided Burdens (Energy Production)		2.01	2.48
Disamenity		-3.65	-7.29
Net Externality (excl. disamenity, incl. displaced burdens from energy generation)		***-6.56***	***-7.69***
Net Externality (excl. disamenity, excl. displaced burdens from energy generation)		***-8.58***	***-10.17***
Net Externality (incl. disamenity, incl. displaced burdens from energy generation)		***-10.21***	***-14.98***
Net Externality (incl. disamenity, excl. displaced burdens from energy generation)		***-12.23***	***-17.46***

Table 19: Externalities of High Quality Landfill in Italy, 5% Discount Rate

Country	ITALY		
Oxidation at Cap			**10.00%**
Landfill Gas Collection Efficiency			**60.00%**
Used to Generate Energy			**50.00%**
Engine Efficiency			**30.00%**
Discount Rate			**5.00%**
		Low	**High**
Global Warming Effects			
	Methane	-1.35	-1.92
	Carbon Dioxide	-0.69	-1.00
Other Air Pollution		-0.03	-0.07
Leachate		-1.00	-2.00
Net External Cost		***-3.08***	***-4.99***
Avoided Burdens (Energy Production)		1.68	2.06
Disamenity		-3.65	-7.29
Net Externality (excl. disamenity, incl. displaced burdens from energy generation)		***-1.40***	***-2.92***
Net Externality (excl. disamenity, excl. displaced burdens from energy generation)		***-3.08***	***-4.99***
Net Externality (incl. disamenity, incl. displaced burdens from energy generation)		***-5.05***	***-10.21***
Net Externality (incl. disamenity, excl. displaced burdens from energy generation)		***-6.73***	***-12.28***

Table 20 Externalities of Lower Quality Landfill in Italy, 1% Discount Rate

Country	ITALY		
Oxidation at Cap			**0.00%**
Landfill Gas Collection Efficiency			**30.00%**
Used to Generate Energy			**30.00%**
Engine Efficiency			**25.00%**
Discount Rate			**1.00%**
		Low	**High**
Global Warming Effects			
	Methane	-11.58	-15.35
	Carbon Dioxide	-6.51	-6.80
Other Air Pollution		-0.08	-0.20
Leachate		-1.00	-2.00
Net External Cost		***-19.18***	***-24.35***
Avoided Burdens (Energy Production)		0.65	0.79
Disamenity		-3.65	-7.29
Net Externality (excl. disamenity, incl. displaced burdens from energy generation)		***-18.53***	***-23.55***
Net Externality (excl. disamenity, excl. displaced burdens from energy generation)		***-19.18***	***-24.35***
Net Externality (incl. disamenity, incl. displaced burdens from energy generation)		***-22.18***	***-30.84***

Net Externality (incl. disamenity, excl. displaced burdens from energy generation)		***-22.83***	***-31.64***

Table 21 Externalities of Top Quality Landfill in Italy, 1% Discount Rate

Country	**ITALY**		
Oxidation at Cap			**50.00%**
Landfill Gas Collection Efficiency			**80.00%**
Used to Generate Energy			**60.00%**
Engine Efficiency			**35.00%**
Discount Rate			**1.00%**
		Low	**High**
Global Warming Effects			
	Methane	-1.65	-2.19
	Carbon Dioxide	-9.51	-9.94
Other Air Pollution		-0.01	-0.03
Leachate		-1.00	-2.00
Net External Cost		***-12.18***	***-14.16***
Avoided Burdens (Energy Production)		4.82	5.93
Disamenity		-3.65	-7.29
Net Externality (excl. disamenity, incl. displaced burdens from energy generation)		***-7.36***	***-8.24***
Net Externality (excl. disamenity, excl. displaced burdens from energy generation)		***-12.18***	***-14.16***
Net Externality (incl. disamenity, incl. displaced burdens from energy generation)		***-11.01***	***-15.53***
Net Externality (incl. disamenity, excl. displaced burdens from energy generation)		***-15.83***	***-21.45***

Landfill performance will vary across countries. However, the Landfill Directive will lead to a degree of standardisation across countries in terms of performance on gas collection and the need to recover energy. As such, over time, one would expect to see some convergence towards good practice. Certainly, it would be incorrect to assume all countries' landfills perform as they do today into the future. For future projections, therefore, it seems reasonable to use the 'best practice' exemplar as an indication of the likely performance of landfills accepting biodegradable waste in future.

Key Results

For the analysis in this work, the key issue is what happens when waste sent to landfill is reduced by one tonne through source separation of materials for composting. In order to understand this, we have effectively calculated the avoided external costs of landfilling waste which is (instead) composted. The assumption is that this material has a composition which is 1% paper, 59% kitchen waste and 40% garden waste. This is different to the typical ratio of separately collected food waste to separately collected kitchen wastes in different countries at present. However, it is a more reasonable estimate of what composition may look like under a requirement for separate collection. The presence of paper reflects the possible use of paper bags to collect the material as well as the possibility of using paper within compost in greater quantities in more remote situations.

The Tables below show the avoided external costs associated with removing one tonne of such material from landfill at different discount rates.

Limitations of the Results

In the general case, we do not have perfect information on a number of key parameters in seeking to model what is going on. To re-emphasise the difficulties in arriving at a 'true' value of the external costs of landfill, we suggest that there will be disagreement about all of the following, each of which determines the external costs of landfilling as we have calculated them:

- Waste composition (varies considerably, each component +/- 50% around the mean, also seasonal);
- Occupational health effects;
- Methane generation by components of landfilled waste (relatively few studies have been done);

- Oxidation rate of methane at the cap (varies with a number of factors – see above);

- Efficiency of landfill gas collection. This is also likely to vary over time for the 'marginal tonne' of waste landfilled;

- Efficiency of engine operation (likely to be better known in specific case, but still exhibiting variation);

- Emissions from displaced energy sources such as one believes the assumptions to be correct. Depending upon one's assumptions, these may be changing, though for a given assumption, the data ought to be reasonably accurate at a given time. However, it is worth pointing out that individual coal plants, for example, differ hugely in their emissions of sulphur dioxide and particulates.

These difficulties are merely those that exist in carrying out the calculations as we have made them. As regards finding a true value, or even a true range, these difficulties are compounded (and one's efforts are confounded) by the various omissions listed above, as well the uncertainties in placing values upon the emissions such as have been quantified. Quantifying the external costs of landfilling is no 'stroll in the park.' Many factors have been omitted. These are:

- Many 'fixed' externalities, such as the impacts associated with landfill construction and engineering, any changes in non-use values of specific sites, and possibly, any non-market benefits from recreational uses post-closure (though these might have to be considered against counterfactual land-uses);

- All impacts associated with the use of on-site vehicles; and

- A number of other impacts whose status is 'unproven' as yet, for example, the possible problems in respect of birth defects that been mentioned in the context of landfilling.

The possibility remains for heavy metals (from, for example, fluorescent tubes) to enter water courses through breaching landfill liners in the future. This is possibly one example of the 'low probability, high consequence risks' which social theorists have recently sought to come to terms with. All of these (apart from the possible benefits from non-market recreation and amenity post-closure – likely to be heavily discounted) are negative externalities. As such, the net externality is a more positive reflection of the true situation than is warranted.

A.3.0 EXTERNAL COSTS AND BENEFITS OF INCINERATION

INTRODUCTION

Again, we lean heavily upon the two studies already discussed, which themselves reviewed a wide range of studies (Broome et al 2000; COWI 2000). The reader is referred to those studies for an overview of the literature. To some extent, the analysis is less problematic than for landfill since emissions are more or less instantaneous. However, some or all of the ash is likely to be landfilled. This will be more or less inert with respect to greenhouse gas emissions, but may not be so in respect of leachate and other environmental impacts. The principal focus is, however, on air emissions and disamenity.

This focus follows in part from earlier work. External costs of air emissions are significant, as are the avoided externalities associated with displacement of fossil fuel energy sources (where this assumption is made). Indeed, the COWI (2000) study bases some of the analysis on the assumption that the plant is a combined heat and power plant. In the Waste Watch study, the greenouse gas emissions are also potentially significant owing to the use of wider ranges of unit damage costs, and inclusion of nitrous oxide emissions in the analysis.

Broome et al (2000) attribute the plant with benefits associated with metals recovery and use of bottom ash to displace virgin aggregates. The COWI (2000) study does not do this. However, it does include an estimate of disamenity which is estimated at £7.50 per tonne. This was based on assumptions that the effect on house prices followed that for landfills, an assumption which is, of course, contentious. However, one might expect significant disamenity impacts. Indeed, the COWI study bases this estimate on housing densities of only 120 houses per square mile, or 47 per square kilometre. This is probably an order of magnitude too low for many municipal waste incinerators, frequently found within, or close to, urban areas.

Both studies took the view that regarding greenhouse gas emissions, the biogenic fraction could be ignored as these emissions would have occurred anyway. This study improves on this analysis, which lacks an important time dimension. All carbon dioxide emissions are reviewed in the analysis, which implies that the analysis is not 'relativised' against any assumed 'emissions that would have occurred anyway'. The reason for this is that the time profile of emissions is potentially important.

THIS ANALYSIS

Air Emissions

For this analysis, we quantify all greenhouse gas emissions. These include emissions of CO_2 in line with Table 22. Nitrous oxide emissions are, unfortunately, not so well related to specific waste fractions (though it may well be that nitrogenous elements such as yard waste, are most responsible for this element).

Table 22 Carbon Dioxide Emissions from Incinerated Fractions

Material	Tonnes CO_2 per tonne Material
Paper/Card	1.47
Yard waste	0.81
Kitchen waste	0.44
Textiles	1.44
Fines	0.53
Timber and Nappies	1.37
Other	0.17
Metals	0.00
Glass	0.00
Plastic	1.91

Notes: All from USEPA (1998) except timber (from BUWAL, cited in RDC and Coopers & Lybrand 1997). Paper / card estimated from several paper types, plastic also estimated on the basis of estimates concerning mixes of polymers. Yard waste is calculated on the basis that this has an average of 50% dry matter, with 75% volatile organic solids (VOS) content and 58% by weight carbon. Kitchen waste assumed to have 25% dry matter, 80% VOS, and 58% carbon.

The N_2O emissions from incinerators were reported by the Intergovernmental Panel on Climate Change (IPCC u.d.). The USEPA (1998) used a figure of 0.01 tonnes metric tonnes of carbon equivalent per tonne MSW combusted. This was based on IPCC reported ranges of N_2O emissions. Because the IPCC did not report N_2O values for combustion of individual components of MSW, USEPA used the 0.01 MTCE figure not only for mixed MSW, but also as a proxy for all components of MSW, except for aluminum and steel cans. The justification for this assumption was that at the relatively low combustion temperatures found in MSW incinerators, most of the nitrogen in N_2O

emissions would be derived from the waste, not from the combustion air. Some N_2O may also be formed in processes, such as SNCR systems, to reduce NOx emissions from the flue. Since aluminum and steel cans do not contain nitrogen, it was assumed that running these metals through an MSW incinerator would not result in N_2O emissions. Such an assumption also appears to support our belief that removing nitrogenous garden wastes from incinerators would also reduce emissions of N_2O, a powerful greenhouse gas (implying that removing such waste from incinerators would be desirable from this perspective). The USEPA (1998) estimate is equivalent to 118kg N_2O per Gg or 0.00012 tonnes N_2O per tonne of waste. This seems reasonable given the range of estimates in Table 23, though Smith et al (2001) used a figure slightly less than half this in their recent report.

Other air emissions are posited to be in line with the new Incineration Directive. This is not strictly consistent with the financial cost analysis since much of the gate fee data will be based upon existing incinerators, some of which may not, in their current form, comply with the limit values in the Directive. Equally, in some countries (for example, the Netherlands, shown in Table 24), emissions limit values for incineration have been tighter, and the financial costs in those countries may reflect these tighter standards. The volume of gas generated per tonne of waste combusted is set at 5 200 Nm3 (studies tend to use a range from 5,000 to 5,500 Nm^3).

Table 23 Emission Factors For N_2O From Waste Incineration

Incineration Plant Type	**MSW**	**Sewage Sludge**	**Hazardous Waste (from industry)**
	kg N_2O/Gg waste (dry)	**kg N_2O/Gg sewage sludge (dry matter)**	**kg N_2O/Gg waste (dry)**
Hearth or grate	5.5-66 (Germany); average 5.5-11; highest value 30 (UK); 40-150 (Japan: wet)	400 (Japan: wet)	NA
Rotating	NA	NA	210-240 (Germany)
Fluidised bed	240-660 (Japan: wet)	800 (Germany); 100-1500 (UK); 300-1530 (Japan: wet)	NA

Note: NA = Not Available.

Sources: Environment Agency (1999), Johnke (1999) and Yasuda (1993).

Table 24 Air Emissions From Incinerators Under New Directive

Emissions (mg/Nm^3)	New EU Directive Limits	Netherlands Limits (Waste Incineration (Air Emissions) Decree
CO_2	-	-
Total dust*	10	5
CO	50	50
SO_2	50	40
NO_2	200	70
HCl	10	10
HF	1	1
Cd + Tl	0.05	-
Hg	0.05	0.05
Cd + Hg	-	
Other heavy metals	0.5 **	1 ***
Dioxins	0.1 ng/Nm^3	0.1ng-
VOCs (TOC)[6]	10	10
Cd		0.05

** Total dust includes particulates (PM_{10})*

*** Sum of Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V, and Sn*

**** Sum of Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V, Sn, Se and Te*

It is important to note that these emissions are not the only ones from incinerators. There are a number of other emissions which occur from incinerators.

Leachate

The degree to which leachate has an impact depends upon what is landfilled and in what quantities. Incinerators produce two forms of ash, bottom ash and fly ash. The

[6] Volatile Organic Compounds (VOCs) are expressed as Total Organic Carbon (TOC) emitted to air.

former, typically 0.2-0.25 tonnes per tonne of waste combusted, is now increasingly used as a material in construction applications. To the extent that this occurs, this material is no longer landfilled. The fly-ash is toxic in nature and may be disposed of at dedicated sites or stabilised before landfilling. Hence, there is typically around 0.05 tonnes of material landfilled per tonne of waste combusted. There have been cases of plant mixing bottom ash and fly-ash for use in construction applications despite what may be elevated dioxin levels in the resulting mixture. This has given rise to considerable disquiet recently given the potential for exposure in subsequent construction and demolition work.

Disamenity

Similar comments apply as were made in the landfill case concerning the fact that the approach typically adopted is to calculate average disamenity values per tonne of waste incinerated over the lifetime of the facility.

The COWI study made the assumption, which is a questionable one (as the study admits), that disamenity could be estimated through assuming the same relationship between house prices and distance from the facility as was assumed for the landfill (based on a meta-analysis of hedonic pricing studies conducted around landfill sites). This was discussed above in the landfill disamenity section.

COWI used a population density around the typical incinerator of 120 per square mile. Yet incinerators are rarely located in such sparsely populated areas, and indeed, where they were, the prospects for district heating would be limited. We would suggest that the population densities are typically an order of magnitude larger. If one uses a figure of 1,200 households per square mile (469 per square kilometre), the values one arrives at for a 200,000 tonnes site, with the annual disamenity being set at 8% of the total, is €75 per tonne. This illustrates how important the population density effect is in this part of the modelling.

In this study, we choose ranges for the household densities likely to be found around incinerators in the different countries. These densities are likely to be at the higher end of the ranges of household density quoted for the countries concerned since currently, the trend has been towards relatively large-scale mass burn facilities. It may be that in future, more modular facilities will be constructed. These are likely to have different emission profiles and costs, as well as disamenity effects since it will be possible to construct them in less densely populated areas. Until then, however, the solution most frequently used is likely to be mass-burn incineration, and such facilities tend to be constructed only where there is a sufficient 'catchment' for waste materials, usually in cities. The figures used are shown in Table 25. They illustrate the pressing need for more thorough research in this area since it is quite clear that the disamenity impacts of such facilities could be extremely significant in the overall external cost estimate for such facilities. At the same time, this is extremely plausible given the typically urban locations of such facilities.

Table 25 Estimated Disamenity Per Tonne (€) From Incineration

	Households per sq mile			Vicinity of Incinerator			
	1998	**1991**		**Assumptions Concerning**		**Disamenity per tonne (€)**	
		Max	*Min*	*Low*	*High*	*Low*	*High*
EU-15	114	0	0				
A	98	0	0	300	1500	18.75	93.75
B	342	6032	54	500	4000	31.25	250.00
DK	143	0	0	350	2000	21.88	125.00
FIN	17	0	0	200	1200	12.50	75.00
FR	106	890	2	300	800	18.75	50.00
GER	256	4326	89	500	3500	31.25	218.75
GRE	66	762	26	300	700	18.75	43.75
IRL	42	0	0	300	1000	18.75	62.50
IT	175	380	33	200	700	12.50	43.75
LUX	161	0	0	400	1500	25.00	93.75
NL	407	1017	112	300	900	18.75	56.25
P	89	263	17	150	250	9.375	15.63
SP	61	3182	16	400	2500	25.00	156.25
SWE	27	0	0	300	800	18.75	50.00
UK	250	4481	0	500	3500	31.25	218.75

Sources: Inhabitants per household from European Environment Agency (1995); population density from Eurostat (2000); COWI (2000); author's calculations.

Avoided Burdens from Energy Production

The approach is as described above. The amount of energy generated is calculated on the basis of the calorific values in Table 22 above and estimates of the efficiency of energy conversion. COWI assumed values of 25% efficiency for incinerators without CHP, and 83% for those with CHP. These values are quite high, especially where incinerators are evaporating effluents from wet scrubbers to avoid discharge of waste effluents. Taking into account own use at the plant, AEA Technology use 18% for electricity generation only and an additional 50% for those also generating heat. In our analysis, we use figures of 21% for incineration without CHP, and 75% for those with CHP. Evidently, the generation of energy per tonne of waste combusted is dependent upon the composition of waste entering the facility. We assume in this study that where the assumption is made that burdens from energy production are being displaced, the displacement effect is pro-rata for the energy generation. As noted above, it could be argued that this ‘over-credits’ CHP plants since the case could be made that the displaced source might be an alternative CHP plant. Given that we have also presented results for no displacement of energy and for the case where only electricity is generated, this is less problematic.

Materials Recovery

The potential for metals recovery post-combustion is something which can, for the most part, occur pre-combustion as well. Indeed, there are benefits in terms of the quality of metal recovered, to extracting the material through source separation of the waste prior to the residual being combusted. In this analysis, since we are not looking at shifts in the management of metals, but in the management of biowaste, we do not attribute any benefit to the incinerator for the metals recovery element. Elsewhere, we have illustrated that where source separation schemes exist, these seem likely to increase the net benefits associated with waste management systems, though proving such a statement conclusively is not without its problems.

Energy from waste incineration plants are increasingly seeking to make use of their bottom ash, often displacing primary aggregate consumption. This is not happening at all plants at present, but there are construction projects making use of bottom ash, and supposing that this practice becomes more widespread in the future, it could be argued that one should include benefits associated with avoided aggregates extraction given that this can follow from the combustion of organic materials.

In a marginalist analysis, it would be wrong, in our view, to simply multiply the mass of bottom ash, typically 0.2-0.25 tonnes per tonne of municipal waste combusted, by the estimated external costs of aggregates extraction, which are taken from work underpinning the UK aggregates tax (London Economics 1999). As discussed in commentaries on the same subject, this estimate is composed of both variable and fixed elements. EFTEC (1999) estimates the variable component of the total as approximately 55%, or 18p per tonne for hard rock outside national parks, £5.79 for hard rock inside national parks, or £1.08 for sand and gravel. In the UK, less than 5% of aggregates come from quarries located in National Parks (and this may fall over time owing to agreements in which operators are engaged). This would imply that for each tonne of municipal waste, then assuming 0.25t of bottom ash used, a net external cost saving of £0.33 (€0.53) in the UK would be the best approximation correct.[7]

In the general Member State case, we do not know the relative proportion of quarrying that occurs for the purposes of sand and gravel, and hard rock extraction, respectively. Hence, it may be wiser to posit a range for the different Member States lying between that implied by the hard rock figure (£0.05, or €0.08 per tonne MSW) and the sand and gravel figure (£0.27, or €0.43 per tonne MSW). We have, in turn, adjusted these to reflect the relative purchasing power of GDP in the different countries.

Note this does no account for any differential transport externalities in transport costs which may arise when one switches from aggregates to bottom ash. Note also that other materials now competing in this market are recycled construction materials. To the extent that one might, at the margin, be replacing secondary aggregates, any additional benefit could (and this is arguable) be reduced to the equivalent of the avoided variable externality associated with secondary aggregates production. Lastly, note that we have not accounted for externalities arising from the removal of contaminants (some of which effectively involves the removal of metals discussed above). Also, heavy metals can be leachable so that in the absence of utilising chemical stabilising agents (at a cost), there may be longer-term effects from the use of bottom ash as substitute for aggregates. These considerations suggest that whilst our analysis suggests a small net benefit, the reality (i.e., if one were able to account for all these impacts) may be rather different (reflecting the fact that this activity is undertaken to avoid the costs of landfilling ash).

Emissions to Water

For completeness, we list the emission limit values for wastewater set out in the Incineration Directive. However, no externality analysis is carried out for the wastewater element.

Incineration plants require permits for discharging wastewater, since the wastewater contains contaminants from Lists I and II of the Dangerous Substances Directive (76/464/EEC). As a result, wastewater is often treated at the incineration plant, particularly at modern plants, prior to discharge to the local sewage treatment plant or

[7] This is calculated as a quarter of the weighted average (by production) externality from the three possible sources assuming no preference for any specific source / type of material. Also, it assumes that bottom ash replaces aggregate on a ‘tonne for tonne’ basis.

direct discharge to surface water. In certain cases, wastewater is evaporated, which transfers the pollutants to the solid and/or gaseous phase. Wastewater treated at the plant can also be reused onsite, which also reduces water consumption.

Table 26 Proposed EU Emission Limit Values (mass concentrations) For Wastewater From Incineration Plants.

Emissions	**Proposed EU Limits** Common Position (2000/C 25/02)**
Total suspended solids*	45 mg/l
Hg	0.03 mg/l
Cd	0.05 mg/l
Tl	0.05 mg/l
As	0.15 mg/l
Pb	0.2mg/l
Cr	0.5 mg/l
Cu	0.5 mg/l
Ni	0.5 mg/l
Zn	1.5 mg/l
Dioxins	0.3 ng/l
Salts (Cl^-, F^-, SO_4^{2-}, PO_4^{3-})	-

** Suspended solids, to which pollutants can partition, cause the water to be cloudy.*
*** Proposed limit values for Total suspended solids, Hg, Cd and Tl may be reduced further; a limit value for total N may be included (25 mg/l); and a pH limit of 6-9 may be included prior to adoption of the Directive.*
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.

RESULTS

As with the landfill case, we take one country (in this case, France) and show external costs of incinerating a tonne of waste of 'standard composition' at 1%, 3% and 5% discount rates (Table 27, Table 28 and Table 29).

Table 27 Externalities from Incinerator in France, 1% Discount Rate

Country				FRANCE
Efficiency of energy conversion (electricity only)				**21%**
Efficiency of energy conversion (CHP)				**75%**
Discount Rate Assumption				**1%**
			Low	**High**
Greenhouse Gases				
	Carbon Dioxide		-41.26	-43.14
	Nitrous Oxide		-2.25	-3.45
Other Air Emissions			-8.72	-23.43
Leachate (landfilled fly-ash)		**0.05**	-0.05	-0.10
Benefits from replaced aggregates		**0.25**	0.07	2.33
Net Externality			*-52.22*	*-67.77*
Avoided Burdens (Energy Production)				
	Electricity		4.24	5.20
	CHP		15.15	18.56
Disamenity			-18.75	-50.00
Net Externality (excl. disamenity, no displaced burdens from energy generation)			-52.22	-67.77
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only)			-47.97	-62.58
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP)			-37.07	-49.21
Net Externality (incl. disamenity, no displaced burdens from energy generation)			***-70.97***	***-117.77***
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only)			***-66.72***	***-112.58***
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP)			***-55.82***	***-99.21***

Table 28 Externalities from Incinerator in France, 3% Discount Rate

Country	**FRANCE**		
Efficiency of energy conversion (electricity only)			**21%**
Efficiency of energy conversion (CHP)			**75%**
Discount Rate Assumption			**3%**
		Low	**High**
Greenhouse Gases			
	Carbon Dioxide	-19.65	-20.69
	Nitrous Oxide	-0.97	-1.68
Other Air Emissions		-8.72	-23.43
Leachate (landfilled fly-ash)	**0.05**	-0.05	-0.10
Benefits from replaced aggregates	**0.25**	0.07	2.33
Net Externality		*-29.33*	*-43.55*
Avoided Burdens (Energy Production)			
	Electricity	4.24	5.20
	CHP	15.15	18.56
Disamenity		-18.75	-50.00
Net Externality (excl. disamenity, no displaced burdens from energy generation)		-29.33	-43.55
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only)		-25.09	-38.36
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP)		-14.18	-24.99
Net Externality (incl. disamenity, no displaced burdens from energy generation)		***-48.08***	***-93.55***
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only)		***-43.84***	***-88.36***
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP)		***-32.93***	***-74.99***

Table 29 Externalities from Incinerator in France, 5% Discount Rate

Country	**FRANCE**			
Efficiency of energy conversion (electricity only)				**21%**
Efficiency of energy conversion (CHP)				**75%**
Discount Rate Assumption				**5%**
			Low	**High**
Greenhouse Gases				
	Carbon Dioxide		-5.37	-7.75
	Nitrous Oxide		-0.25	-0.36
Other Air Emissions			-8.72	-23.43
Leachate (landfilled fly-ash)		**0.05**	-0.05	-0.10
Benefits from replaced aggregates		**0.25**	0.07	2.33
Net Externality			*-14.32*	*-29.30*
Avoided Burdens (Energy Production)				
	Electricity		4.24	5.20
	CHP		15.15	18.56
Disamenity			-18.75	-50.00
Net Externality (excl. disamenity, no displaced burdens from energy generation)			-14.32	-29.30
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only)			-10.08	-24.10
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP)			0.83	-10.74
Net Externality (incl. disamenity, no displaced burdens from energy generation)			***-33.07***	***-79.30***
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only)			***-28.83***	***-74.10***
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP)			***-17.92***	***-60.74***

The results show a similar effect (as with landfill) of discounting on global warming externalities. The obvious fact is that how one perceives the 'displaced energy' question plays an important role in the analysis.

For comparison, we show below the results for the UK (Table 30, Table 31 and Table 32). The things that change are:

- The non-global warming air pollution externalities (since some of these have country-specific unit damage costs applied to them);
- The displaced burdens from energy generation, where one accepts that this is the correct way to treat the issue; and
- The disamenity (related to population densities).

The first of these is relatively uncontentious. The enormous variation which results from the second and third factors suggests the need for considerable caution in interpreting the results.

The results appear to suggest that as the discount rate increases, so global warming externalities fall whilst 'displaced burdens' remain constant. Hence, in the UK case, where one accounts for displacement of energy related externalities, the higher the discount rate applied, the more 'positive' the picture regarding incineration appears. This is a problem in this analysis since the displaced energy externalities are not varying with the discount rate as are the externalities associated with greenhouse gases from incinerators.

Furthermore, it should be pointed out that the avoided burdens from energy generation are from relatively dated studies, whilst for the United Kingdom, for example, not all incinerators are yet meeting the emissions standards set under the Incineration Directive. In future, one would expect 'displaced burdens', to the extent that one believes this approach to be the correct one, to become much smaller as a) improvements are made in emissions abatement and b) a greater proportion of energy supply is from renewable resources. To that end, the energy displacement effects depicted in the analysis are likely to tend more towards the situation depicted in the French analysis than in the UK one.

These comments help to show how contentious the treatment of 'displaced burdens' can be since somewhat perversely, the external benefits attributed to incinerators are enhanced by the use of more polluting fuels elsewhere in the economy where one assumes that emissions from these are 'being displaced'.

Table 30 Externalities from Incinerator in United Kingdom, 1% Discount Rate

Country	UK			
Efficiency of energy conversion (electricity only)				**21%**
Efficiency of energy conversion (CHP)				**75%**
Discount Rate Assumption				**1%**
			Low	**High**
Greenhouse Gases				
	Carbon Dioxide		-41.26	-43.14
	Nitrous Oxide		-2.25	-3.45
Other Air Emissions			-8.44	-20.28
Leachate (landfilled fly-ash)		**0.05**	-0.05	-0.10
Benefits from replaced aggregates		**0.25**	0.07	2.41
Net Externality			*-51.93*	*-64.55*
Avoided Burdens (Energy Production)				
	Electricity		25.77	32.40
	CHP		92.02	115.73
Disamenity			-31.25	-218.75
Net Externality (excl. disamenity, no displaced burdens from energy generation)			-51.93	-64.55
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only)			-26.16	-32.15
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP)			40.10	51.18
Net Externality (incl. disamenity, no displaced burdens from energy generation)			***-83.18***	***-283.30***
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only)			***-57.41***	***-250.90***
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP)			***8.85***	***-167.57***

Table 31 Externalities from Incinerator in United Kingdom, 3% Discount Rate

Country	**UK**			
Efficiency of energy conversion (electricity only)				**21%**
Efficiency of energy conversion (CHP)				**75%**
Discount Rate Assumption				**3%**
			Low	**High**
Greenhouse Gases				
	Carbon Dioxide		-19.65	-20.69
	Nitrous Oxide		-0.97	-1.68
Other Air Emissions			-8.44	-20.28
Leachate (landfilled fly-ash)		**0.05**	-0.05	-0.10
Benefits from replaced aggregates		**0.25**	0.07	2.41
Net Externality			*-29.04*	*-40.33*
Avoided Burdens (Energy Production)				
	Electricity		25.77	32.40
	CHP		92.02	115.73
Disamenity			-31.25	-218.75
Net Externality (excl. disamenity, no displaced burdens from energy generation)			-29.04	-40.33
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only)			-3.28	-7.93
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP)			62.98	75.40
Net Externality (incl. disamenity, no displaced burdens from energy generation)			***-60.29***	***-259.08***
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only)			***-34.53***	***-226.68***
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP)			***31.73***	***-143.35***

Table 32 Externalities from Incinerator in United Kingdom, 5% Discount Rate

Country	**UK**			
Efficiency of energy conversion (electricity only)				**21%**
Efficiency of energy conversion (CHP)				**75%**
Discount Rate Assumption				**5%**
			Low	**High**
Greenhouse Gases				
	Carbon Dioxide		-5.37	-7.75
	Nitrous Oxide		-0.25	-0.36
Other Air Emissions			-8.44	-20.28
Leachate (landfilled fly-ash)		**0.05**	-0.05	-0.10
Benefits from replaced aggregates		**0.25**	0.07	2.41
Net Externality			*-14.03*	*-26.08*
Avoided Burdens (Energy Production)				
	Electricity		25.77	32.40
	CHP		92.02	115.73
Disamenity			-31.25	-218.75
Net Externality (excl. disamenity, no displaced burdens from energy generation)			-14.03	-26.08
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only)			11.73	6.32
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP)			77.99	89.65
Net Externality (incl. disamenity, no displaced burdens from energy generation)			***-45.28***	***-244.83***
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only)			***-19.52***	***-212.43***
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP)			***46.74***	***-129.10***

Limitations of the Results

It should be re-stated that this is a far from complete analysis. The following impacts have not been covered: [8]

- All emissions to water;
- Some air emissions for which no externality adders were available;
- Fuel use associated with on-site vehicles;
- Impacts associated with extracting and cleaning bottom ash recovered for use in construction, and transporting this to users;
- Transport externalities associated with landfilling of fly-ash;
- Extraction of primary resources (such as lime, which is often used in cleaning flue gas, and water);
- Occupational health effects; and
- 'Fixed' disamenities associated with plant construction.

As with the landfill case, we do not have very good information on a number of key parameters in seeking to model what is going on. Also as in the landfill case, all the unquantified externalities are negative ones. Hence, the net figure is not an accurate reflection of the true situation, which would ideally incorporate the negative externalities mentioned. To re-emphasise the difficulties in arriving at a 'true' value of the external costs of incineration, we suggest that there will be disagreement about all of the following, each of which determines the external costs of incineration as we have calculated them:

[8] Many studies list several waste materials, emissions to air not covered by us, and other residues arising from incineration of municipal waste (see, for example, Kremer et al 1998).

- Waste composition (varies considerably, each component +/- 50% around the mean, also seasonal);

- An exact computation of the links between waste components and emissions to different media. Data were used to link CO_2 emissions to specific components of the waste stream. However, in the general case, a number of factors will affect emissions from incinerators (inputs by composition, but also by quantity, depending on how the incinerator has been specified) including the nature and performance of flue-gas cleaning equipment;

- The relevance or otherwise of less frequent exposures to higher emissions of specific pollutants in determining ultimate effects upon which externality calculations are based;

- Efficiency of energy recovery (likely to be known for certain conditions in a specific case, but still exhibiting variation across plants and according to, e.g., completeness of combustion); and

- Assumptions concerning so-called displaced energy sources'.

Again as with our landfill, these difficulties are merely those that exist in carrying out the calculations as we have made them. Finding a true value, or even a true range, is made very difficult indeed by the various omissions listed above, as well the uncertainties in placing values upon the emissions such as have been quantified.

Lastly, there has been no account taken of the possibility of technical failure / human error / mismanagement leading to consequences for health and the environment. For example, considerable concern has arisen in the UK recently from incidents of mixing of fly-ash and bottom ash residues, these having been spread (in at least one case) on allotments used to grow vegetables. The consequences in terms of dioxin exposure are allegedly quite serious. Potential risks associated with mal-function / mismanagement are shown in Table 33.

Table 33 The Main Risks Of Accidental And Sudden Occurrences At Incineration Plants.

Potential risk	Description and consequences
Contact with auxiliary materials	Occupational health and safety risk to workers at the incineration plant. Risk of dermal contact, and associated health effects.
Fire in the silo	Occurs relatively frequently and is extinguished using water canons permanently installed in the silo. Causes an increase in emissions to air, until fire is extinguished.
Fire in the dioxin filter	May occur if glowing particles reach the filter. Results in fine contaminated carbon particles being emitted to air (e.g. in the order of a few tonnes), requiring the installation of a new filter.
Leaks from high pressure feed-water and steam system	Occupational health and safety risk to workers at the incineration plant. Risk of dermal contact and associated health effects.
Overheating	Overheating of the boiler may cause damage and outlets of steam in the boiler. Results in temporary reduction or stop in plant operation and energy recovery.
Explosive matter in the waste	Waste may contain explosive matter such as gas bottles that can explode in the boiler, silo, or shredder. Explosions can result in temporary reduction or stop in plant operation and energy recovery.
Leak of ammonia	A sudden leak of ammonia, used to remove NOx, is acutely dangerous. However, even small leaks are easily detected by smell long before dangerous concentrations are reached. The main risks relate to occupational health and safety.
Contact with flue gas residues	Occupational health and safety risk to workers at the incineration plant. Risk includes dermal contact and inhalation of fugitive dust emissions, and associated health effects.

Source: COWI (2000).

A.4.0 APPENDIX FOUR EXTERNAL COSTS AND BENEFITS OF COMPOSTING

INTRODUCTION

Full life-cycle analyses of composting processes are difficult to carry out. Furthermore, life-cycle approaches are unlikely to capture all of the processes which an attempt to derive external cost estimates should seek to cover. One of the difficulties associated with the approach to life-cycle analysis regarding composting is the variety of processes available and the fact that different processes make use of different feedstocks. Ideally, as with all treatments, one seeks to relate specific emissions and outputs to the input materials (although in practice, this is difficult to achieve, and one suspects that especially with biological processes, synergistic effects may occur). This has only been attempted seriously in one of the models we have looked at.

There are only a relatively small number of studies that appear to have attempted to look at the emissions from composting plants. They do not all deal with the same materials. They are also carried out in different technological processes. The models we have looked at are:

- IWM and IWM2 life-cycle models (White et al 1995; 2000);
- UK Environment Agency data (used in the development of the WISARD life cycle tool) (Environment Agency 2000);
- ORWARE model (developed in Sweden) (Sonneson 1997);
- A Dutch study (de Groot and van Lierop 1999); and
- A review of studies undertaken by Aumonier (1999).

On closer examination, each of these appears to have problems associated with it. In some cases, the errors are basic calculation errors, and they may have a profound effect on the modeling. There is a clear need for improved data (and transparency in the attempt to convey what the data applies to) with attempts made to relate emissions in the process to input materials. Only the ORWARE model appears to have made such an attempt.

EMISSIONS DATA FOR COMPOST

Gaseous Emissions Data

On gaseous emissions, the White et al. (IWM) model only has 'its own' data for CO_2. For other emissions, the model uses emissions from the ORWARE model. The value of 320,000 g/tonne MSW is consistent with the data taken from the IWM 2 model even though a factor of 10,000 appears to have been omitted (as it is only quoted as 32 g CO_2 per tonne in IWM2). In the actual IWM2 model, all emissions except for CO_2 are completely ignored. From examining the IWM2 model, the data appears to be for the process only so there is no attempt to account for sequestration of biogenic carbon through compost application, nor any attempt to understand the potential for pollution displacement from replacement of other products. This is a very common shortcoming in this type of analysis. As such, one can state that, to the extent that avoided burdens associated with energy recovery have tended to be assigned to energy recovery options, comparative assessments have been biased against compost (since it does not recover energy, but displaces, instead, burdens that have rarely been quantified in anything like a serious manner).

Considerable problems were experienced interpreting the data presented in the UK Environment Agency (EA) model. The lack of transparency and clarity in the presentation of makes the report extremely awkward to interpret. The report gives emissions data for two plants (Dogsthorpe and Cambridge, Outdoor and Covered respectively). There is no indication as to how this data is derived though there are hints that the data for Dogsthorpe is derived from that for Cambridge (though we are unsure as to how this was accomplished).

The figures for CO_2 emissions appear to include energy use from the use of plant machinery and possibly also from construction, as well as emissions related to the composting process itself. There is a calculation for the total CO_2 emitted in the 'process only' for Dogsthorpe (387,053 g/tonne CO_2) which is a different figure to that quoted in the table for (presumably) total emissions from Dogsthorpe (482,000 g/tonne). This would indicate that all the data in the EA Study column is not just from the process (for both composting sites). This is supported by the fact that the EA Study provides the only non-zero data for particulates from composting which (although it seems to be acknowledged that they do exist near the process 'naturally') indicates combustion and therefore probably includes energy use from on site machinery, but possibly also, construction of plant. The data for CO_2 emissions per tonne MSW

(387,053 g) from the process is broadly in line with that from the White et al study discussed above. Since we have accounted for energy separately, the particulate emissions are ignored although it is clear that there will be dust arising from composting, particularly in the curing phase of operation.

The Dutch study does not provide detailed emissions data. The study states that the process involves 87kg aerobic digestion per tonne of MSW composted. If all of this mass loss was carbon, we calculate that this would give rise to emissions of 331,760g CO_2 per tonne MSW. This estimate is in line with the other figures quoted.

The last of the models to be described in detail is the ORWARE model. It gives rise to the most complete set of data for composting emissions. This model makes a number of assumptions, for example, that the gaseous emissions from composting are the same irrespective of the process involved (windrow, reactor, and home), and that the emissions from different components of waste can be modelled discretely independent of the mix if materials involved. The energy consumption varies according to process however.

Emissions from the process vary according to the feedstock type. No other models have attempted to model this variation. The model provides detailed emissions data depending on the feedstock composition. Using a detailed composition of source separated organic waste used as an example in the ORWARE model, we calculated the emissions from the composting process. The composition of waste used in this calculation is shown in Table 34.

Table 34 Composition Of Source Separated Waste Used In The ORWARE Model Calculation (kg/kg waste)

Material	Fraction	Material	Fraction
C-Total	0.434	C/N Ratio	21.7
C-Lignin	0.029	VOC	2E-06
C-Cellulose	0.107	CHX	1E-08
C-Starch	0.097	PAH	5E-07
C-Fat	0.135	Phenols	3E-05
C-Protein	0.066	PCB	4E-08
N-Total	0.02	Dioxin	9E-14

Source: Based on Sonneson 1997

Using this data, the model calculates the emissions of a number of species (CO_2, N_2O, N_2, NH_3, VOC's, PCB's) using factors associated with the proportion of each species in the incoming material which degrade giving rise to gaseous emissions. The emissions of CH_4 are then calculated by assuming a set percentage of the CO_2 produced is CH_4.

The findings of this study using the ORWARE model can be compared with other studies including White et al. which quotes results using the ORWARE model. The calculated emissions of CO_2 equal 562,000 g per tonne of waste treated. This is very similar to the White et al. quoted results of the ORWARE study (566,000 g per tonne). Data for emissions of the rest of the species calculated in this study with the ORWARE model can also be compared with those quoted in White et al. These are also very similar indicating that it is likely that the model has been used in a consistent manner.

The ORWARE model also provides data for the emissions associated with the composting process with gas cleaning. According to the results from the model, this process reduces emissions from some of the species (CH_4, NH_3, N_2O) by around 90%. From the model description the main process of emissions reduction seems to be filtration through the compost itself. This has the advantage of not simply transferring the pollution to another media which is a problem associated with gas scrubbing technologies in general.

The emissions data examined above are associated only with the composting process. Obviously in addition to these emissions, there are also those from energy use which need to be factored in. The IWM2 model, while not providing detailed emissions from the composting process, provides emissions data for the energy use on site. These issues are described below.

It seems clear that the most detailed data on emissions from the composting process available at present are from the ORWARE model. On the other hand, their reliability is called into question by the fact that they are very much higher than other studies referenced. One of the problems may be the assumption that the feedstock is typically 500kg dry matter per tonne. This is very high given that the moisture content of kitchen wastes in particular is considerable (perhaps as high as 80%). A dry matter content of 35% or so would make the carbon dioxide emissions comparable across all studies. Composition also affects C/N ratios of input material, which are used as a parameter for modelling N-losses within the model. The modellers recognise that the fate of nitrogenous emissions is an area in need of improvement. In general, higher C/N ratios are deemed to lead to lower N losses.

A simple mass balance would suggest that, if the input material has a moisture content of 70%, and if the volatile solids content of the remainder is 80%, then input waste contains 240kg of volatile solids. Unscreened stabilised biomass might be 40% of the initial weight, of which 40% might be moisture with the remainder having a volatile solids content of 40%. Ths implies a loss of 144kg volatile solids if the incoming material has a moisture content of 70%. If all carbon in the lost volatile solids is converted to carbon dioxide, this implies emissions of 306,225g. At 50% moisture content (as assumed by ORWARE) some 646,558g would be emitted using the same calculation, which is tolerably close to the ORWARE figure, especially if one considers that ORWARE suggests the emission of some methane. This illustrates the importance of the role of waste composition on emissions of carbon dioxide from compost.

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.

Table 35 below shows the emissions data from the studies discussed and those carried forward in the study. Note that for all emissions other than carbon dioxide, the lower end of the range is likely to be a better reflection of the situation as regards the more advanced compost plants deploying biofilters to reduce gaseous emissions. This is modelled explicitly in the ORWARE study. Given that we have been asked to assess good practice in composting, then since this should be reflected in well-controlled processes, the emissions data carried forward tends to reflect an assumption of good process controls (and effectively, this is what we have also done for landfills and incinerators). The most interesting emissions category from this perspective may be volatile organic compounds (VOCs). VOC's may need to be abated at mechanical biological treatment (MBT) sites, as mixed waste contains some potentially hazardous VOC's (paintings, solvents, etc.). The German Government has, for instance, set a limit value for overall VOC's emissions at 55 grams/ton. This applies for the time being only to MBT facilities. At composting facilities, most VOC's are probably being produced by the biofilter itself in a natural way, as e.g. terpenes come from degradation of wooden materials of the biofiltering media. This is why VOC elimination may be unlikely. The attempt to clean exhaust gases itself leads to emissions of some VOCs. Best practice would be expected, therefore, lead to emissions of VOCs. Interestingly, the ORWARE model suggests a complete removal of these through gas cleaning which may be unrealistic.

Table 35 Gaseous Emissions Data From Composting Process (all figures are g per tonne of waste composted)

Gas	White / IWM	EA -Study[9]		Aumonier	Dutch	ORWARE		Stats Used	
		Outdoor	Covered			Without gas cleaning	With gas cleaning	Min	Max
CO_2	320,000	387,000	426,000	320,000 (566,000)	331,760	561,917	562,900	350,000	
CH_4				2,000		1,967	983	983	
N_2O				97		110	11	11	
Dioxins/furans						0	0	0	
Ammonia		60	5	3,000		3,706	371	371	
VOCs		24				0.7		24	
Particulates		186	163					0	

Energy Related Emissions Data

The energy used at the plant also contributes to the emissions associated with the composting process. Estimating the energy consumption from a composting process in the general sense can be difficult as it requires assumptions as to the type of process involved. As stated above, while direct emissions from the different process types may remain similar (although even this is a questionable assumption), the energy consumption will vary considerably. This is borne out by the findings from of our survey of existing energy consumption estimations.

In general energy consumption varies according to process type. Windrow systems seem to use less electricity and more liquid fuel while reactor systems use more electricity and less fuel. Therefore, simply deriving ranges using lower and upper bound estimations for both electricity and fuel use is likely overestimate the range of total energy consumption.

ORWARE assume 97kJ/kg (27kWh per tonne) electricity and 5kJ/kg diesel oil for enclosed composting whilst the assumption for windrows is 0kJ/kg electricity and 15kJ/kg diesel oil. An additional allowance for pre-treatment where material is collected

[9] It is impossible to tell from the Environment Agency study where the estimate for particulates comes from. It may be that this is associated with fuel used, but since we treat this separately, this is not used in the analysis.

in plastic bags can be allowed for, but in our view, where plastic bags are used, they should be of the quickly biodegrading type (e.g. corn starch).

The plant in Middenmeer in Netherlands is reported as having a consumption of electricity of 260 MJ/tonne (72kWh per tonne). The UK Environment Agency study seems to suggest 4l of diesel per tonne of waste input in open windrow composting.

This study has explored the issue of best practice with respect to composting technologies. We estimate that energy use amounts to approximately 50kWh per tonne and 1 litre of fuel (use of loaders). The treatment of the 50kWh can be dealt with in the externality analysis in the same way as the avoided energy emissions (which is discussed above). 1 litre of fuel would generates the emissions shown in Table 36.

Note that the balance between fuel and electricity may itself be considered in due course as an environmental factor in production. The use of loaders might be eliminated further in best practice plants where conveyors are used as the prinicipal means of material transport.

Table 36 Emissions From Diesel Engine (per litre)

Air Emissions, mg	**g/l**
Particulates	2.564
CO	26.548
CO2	3036.258
CH4	0.336
Nox	33.901
N2O	0.041
Sox	10.106
HCl	0.038
HF	0.038
HC	10.898
Water Emissions, mg	
BOD	0.038
COD	0.038
Susp. Solids	0.038
TOC	0.415
Phenol	0.038
Total metals	0.038
Cl	0.038
F	0.038

Source: White et al. (1995).

BENEFITS FROM COMPOST UTILISATION

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.

For example, 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, has 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, as Freckman (1994) puts it, '*Soils are one of the most poorly researched habitats on earth*'. There are interactions occuring, therefore, that are only relatively poorly understood.

Agricultural Applications

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.

Table 37 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.

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 especially relevant to countries where the soils have a low organic matter content, as is common in Mediterranean countries. Montanarella (1999) estimated that over 150 million hectares of European soils are suffering from erosion, the problem being more acute in southern countries. Soil erosion is a major socio-economic and environmental problem throughout Europe. It reduces the productivity of the land and degrades the performance and effectiveness of ecosystems.

Table 37 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 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 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. Furthermore, organic farming certification bodies are increasingly wary of using composts derived from municipal waste since they are concerned at the potential for contamination with genetically modified materials, specifically, seeds which may still be capable of germination after the composting process.

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' 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 (for example, where they are required to carry out mineral balances, as, for example, in the Netherlands and parts of Finland). We therefore use 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. This is equivalent to approximately 16.7 tonnes of compost, derived from approximately 47.7 tonnes of waste material (assumes 1 tonne of waste material leads to production of 350kg compost with dry matter content 60%). We assume further that this material has the following composition in terms of nutrients:

Nitrogen:	1.5% dry matter
Phosphorous (as P_2O5):	1.0% dry matter
Potassium (as K_2O):	1.2% dry matter

These figures are representative of those quoted in various sources. Table 38 from VLACO’s programme of testing for compost quality shows the variation in nutrient content across the three types of compost (biocompost, humotex and green comost). The biocompost, from biowastes, shows higher nutrient content in all respects. A Table from FEAD’s investigations is shown in Table 39. Our figures for P_2O5 and K_2O are slightly higher. This is because we assume that the material has a significant kitchen waste fraction, which is likely to increase the nutrient content of the compost. The figures below are likely to include composts derived principally from garden waste.

Table 38 Nutrient Content of Different Composts

	Biocompost	Humotex	Green compost
Dry Matter	66.70%	53.80%	58%
Moisture	33.30%	46.20%	42%
N tot fresh	1.16	0.77	0.7
N tot dry	1.78	1.48	1.2

Total P_2O_5 (% weight)	0.61	0.46	0.3
Total P_2O_5 (% dry matter)	0.91	0.86	0.52
Total K_2O (% weight)	0.91	0.38	0.54
Total K_2O (% dry matter)	1.36	0.71	0.93
Total CaO	2.13	2.07	1.42
Total MgO	0.39	0.33	0.25

Source: VLACO (2000)

Table 39 Nutrient Content Of Composts Reported In Literature

	Organic Substance		**Nitrogen (N)**		**Phosphate (P)**		**Potassium (K)**		**Number of Plants in Sample**
	average % d.m.	**range % d.m.**	**average % d.m.**	**range % d.m.**	**average % d.m.**	**range % d.m.**	**average % d.m.**	**range % d.m.**	
Denmark		18-50							10
France	44		1.5		0.5		0.8		3
Germany	36	24-51	1.3	0.8-1.9	0.26	0.17-0.48	0.91	0.5-1.41	1500
Luxembourg	?	?	2.09	1.94-2.19	1.35	1.08-1.51	1.97	1.64-2.19	2
Netherlands	38		1.59		0.29		0.88		88
Sweden	56	45-81	2.4	1.3-4.2	0.45	0.27-0.56	1.13	0.88-1.5	5
UK	19	11.6-26.3	1.2	0.94-1.8	0.18	0.15-0.26	0.63	0.4-0.84	5

The mineralisation rate of the nutrients is assumed to be 30% for all nutrients in Southern Member States, 25% in France and 20% in Northern Member States. This determines the time profile of the displacement effect (which in turn affects the external benefits associated with dispacement via the discounting mechanism).

For synthetic fertilisers, a loss rate of 23% is assumed for nitrogenous fertilisers.[10] 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 Table 40 and Figure Y below. Equivalent projections for P and K displacement are given in

[10] This is the loss rate from Hydro Agri Europe (1995) *Important Questions on Fertilizer and the Environment*, Brussels: Hydro Agri Europe.

Table 41 and Table 42.

Table 40: 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	58.4	**58.4**
2	40.9	**99.4**
3	28.6	**128.0**
4	20.0	**148.0**
5	14.0	**162.1**
6	9.8	**171.9**
7	6.9	**178.8**
8	4.8	**183.6**
9	3.4	**186.9**
10	2.4	**189.3**

Figure 2: Evolution of Nitrogen Displacement Over Time from Single Compost Application

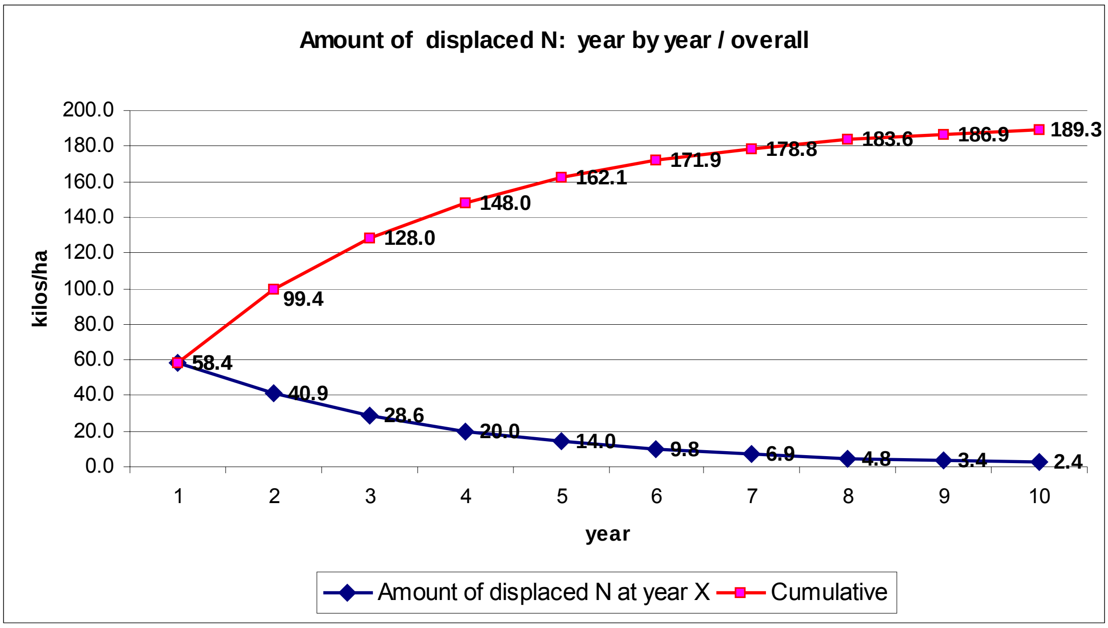


Table 41: Evolution In P_2O_5 Displacement Associated With 10 Tonnes Dry Matter Of Compost Applied To Farmland, Southern Member State Case

Year	P_2O_5 Displacement (kg)	Cumulative (kg)
1	15.0	15.0
2	10.5	25.5
3	7.4	32.9
4	5.1	38.0
5	3.6	41.6
6	2.5	44.1
7	1.8	45.9
8	1.2	47.1
9	0.9	48.0
10	0.6	48.6

Table 42: Evolution In K_2O Displacement Associated With 10 Tonnes Dry Matter Of Compost Applied To Farmland, Southern Member State Case

Year	K_2O Displacement (kg)	Cumulative (kg)
1	36.0	36.0
2	25.2	61.2
3	17.6	78.8
4	12.3	91.2
5	8.6	99.8
6	6.1	105.9
7	4.2	110.1
8	3.0	113.1
9	2.1	115.2
10	1.5	116.6

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.

Fertilisers vary a great deal in terms of nutrient content and therefore the externalities will vary also. This study used data for the production of NPK fertiliser because this is the most widespread in terms of use, and the most readily available from a data perspective. Within this fertiliser type there are a number of different varieties depending on the comparative contents of N:P:K nutrients. Again because of data availability, this study calculated the externalities associated with the production of one tonne of 15:15:15 NPK fertiliser (i.e. 150kg N, 150kg K_2O and 150kg P_2O_5).

There are a number of different routes to NPK fertiliser production. The most widespread are the mixed acid and nitrophosphate routes. This study used data on Best Available Technologies from EFMA (European Fertilisers Manufacturing Association). Most of the data available 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. Therefore these assumptions will incur errors. However, these are unlikely to be significant compared to other areas.

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 K2O portion and 150kg P_2O_5. 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).

We have not been able to obtain quality data for extractive processes. 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_2O_5) (100%) (Bocoum and Labys 1993). Energy use for producing phosphate rock has been estimated at 73.5 kWh/tonne (UNEP and UNIDO 1998). Additional energy consumption for phosphate fertiliser is attributed on this basis.

Because we have no information concerning emissions in potash production, the emissions data for K_2O are likely to understate the environmental benefits of displacement effects. We have obtained a figure for CO2 emissions and have used the best practice figure from this (Kongshaug 1998). It is important to note that these externalities are associated with Best Available Technologies for both new and existing plants. Therefore the costs in terms of gaseous emissions and energy requirements are likely to be underestimated as not all plants will be using Best Available Technologies. Technological improvements in the future however are likely to mean that these levels will become more similar to emission levels observed in practice. The emissions, and energy data used, are shown in Table 43.

Table 43 Emissions Data Used For Fertiliser Manufacture

Tonnes emission/kg nutrient	N		P_2O_5		K_2O	
	Low	*High*	*Low*	*High*	*Low*	*High*
CO_2	2.565E-03	2.565E-03	0	0	1.3E-04	1.3E-04
NO_x (as NO_2)	4.087E-06	2.143E-05	6.67E-07	6.67E-07	6.67E-07	6.67E-07
N_2O	9.28E-06	3.519E-05	0	0	0	0
NH_3 (as N)	4.444E-07	4.444E-07	4.44E-07	4.44E-07	4.44E-07	4.44E-07
Fluoride (as F)	4.444E-08	4.444E-08	7.11E-08	2.04E-07	4.44E-08	4.44E-08
Dust/particulates	4.444E-07	4.444E-07	7.11E-07	1.24E-06	4.44E-07	4.44E-07
SO_2	4.373E-07	1.441E-06	4.3E-07	1.43E-06	4.3E-07	1.43E-06
SO_3	2.15E-08	0.000000086	2.15E-08	8.6E-08	2.15E-08	8.6E-08
CO	1.29E-08	1.29E-08	0	0	0	0
Energy Use (drying) (MJ)	5.776E-04	0.000577593	0.000578	0.000578	0.000578	0.000578
Elec Use (kWh)	8.3704E-05	8.37037E-05	0.000482	0.000482	8.37E-05	8.37E-05

As a comparison, the energy savings estimated by one study from the use of compost were estimated at 100kWh per tonne of waste. Similarly, another study suggested that the production of nitrogen fertilisers (commonly ammonium nitrate) requires large quantities of energy (in particular natural gas), both as an energy feedstock and for fixation of atmospheric nitrogen. In order to produce one tonne of nitrogen fertiliser, a modern process produces 2.2 tonnes of carbon dioxide (CO_2) for every tonne of nitrogen fertiliser produced (IFA 1997). Manufacture of nitrogen fertiliser was also estimated to lead to emissions of nitrous oxide (N_2O) equivalent to £0.56 per tonne of nitrogen fertiliser (Soulsby et al 2000). Based on the assumptions made and for the environmental impacts identified, the total external cost was estimated at £14.33 per tonne of nitrogen fertiliser product used in the UK.

Phosphate Fertiliser - Phosphogypsum and Process Wastewater Disposal

Soulsby et al (2000) 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. 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_2O_5 in phosphoric acid of 32% and a concentration of 46% of P_2O_5 in the product. This would be equivalent to £59.91 (€95.86) per tonne P_2O_5, or €0.096 per kg P_2O_5. We have used this as an admittedly crude estimate of the externalities associated with P_2O_5 production (which may be avoided when compost is applied).

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 denitrificaation, 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_20. 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 nitrate appear to lose much less nitrogen in this way (Lashof and Tirpak 1990; Ehrlich 1990). 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 (McTaggart et al 1998). A recent Dutch study cites figures for N2O losses as between 1 and 3% of mineral N applied (Mosier 1993).

The following nitrogen volatilisation rates have been derived using the MANNER Model[11]. 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_3-N and NH_4-N).[12]

<u>External Costs</u>

The valuation of reduced N2O 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 N2O 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_2O. 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 issue also depends upon the mineralisation rate, which is, in our study, country specific.

For the 3% Discount Rate case, and for Southern Member State mineralisation rate, the results are shown in

[11] 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.
[12] Figure calculated using data from HDRA Consultants Compost Analysis Database which contains data from over 80 different composts.

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;

- Successful competition for nutrients by beneficial micro-organisms.
- Antibiotic production by beneficial micro-organisms.
- Successful predation against pathogens by beneficial micro-organisms.
- Activation of disease resistant genes in plants by composts.

Table 44 expressed per tonne of waste composted. The benefits range from €0.24 - €1.62, a range which arises from combining low emissions with low unit damage costs, and high emissions with high unit damage costs. The assumption in the Table is that the nitrogen is replaced on a one-for-one basis. This is amended in the final calculations.

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- Activation of disease resistant genes in plants by composts.

Table 44 External Benefits From Reduction In N_2O Emissions Through Replacing N Fertiliser With Compost

N replaced by 10	**N_2O emissions @ 0.05%**	**N_2O emissions @**	**External Benefit (€**	**Externa Benefit (€**

tonnes compost (kg)	(tonnes, low)	0.5% (tonnes, high)	per tonne waste composted, Low)	per tonne waste composted, High)
58.44	9.64E-06	3.86E-05	0.08	0.55
40.91	6.75E-06	2.70E-05	0.06	0.38
28.64	4.73E-06	1.89E-05	0.04	0.27
20.05	3.31E-06	1.32E-05	0.03	0.19
14.03	2.32E-06	9.26E-06	0.02	0.13
9.82	1.62E-06	6.48E-06	0.01	0.09
6.88	1.13E-06	4.54E-06	0.01	0.06
4.81	7.94E-07	3.18E-06	0.01	0.04
3.37	5.56E-07	2.22E-06	0.00	0.03
2.36	3.89E-07	1.56E-06	0.00	0.02
1.65	2.72E-07	1.09E-06	0.00	0.02
1.16	1.91E-07	7.63E-07	0.00	0.01
0.81	1.33E-07	5.34E-07	0.00	0.01
0.57	9.34E-08	3.74E-07	0.00	0.01
0.40	6.54E-08	2.62E-07	0.00	0.00
0.28	4.58E-08	1.83E-07	0.00	0.00
0.19	3.20E-08	1.28E-07	0.00	0.00
0.14	2.24E-08	8.97E-08	0.00	0.00
0.10	1.57E-08	6.28E-08	0.00	0.00
0.07	1.10E-08	4.40E-08	0.00	0.00
		NPV@3%	£0.24	£1.62

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. Supressiveness is not a consistent quality, therefore to accommodate this, the compost feedstock should be as homogenous and as consistent as possible. Homogenicity 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. Supressiveness 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 (for a review, see Hoitink and Boehm 1999).

In addition, several studies have shown soil microbial activity to be higher, and disease incidence lower, after the application of compost (1996, Costa *et al*; 1997, Kim *et al*). The following table is a summary of specific work carried out on specific plant pathogens. In these studies, various degrees of control were achieved by using compost.

Table 45 Research into control of plant pathogens by the addition of compost

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

Sources: Hoitink, H.A.J. and Fahy, P.C. (1986); Ringer et al. (1997) and Yohalem et al. (1994).

External Costs of Pesticides

A number of attempts have been made to estimate in money terms the environmental costs of using pesticides.[13] Pimentel et al (1993) estimate the total external costs of pesticide use in US agriculture at just under $8 billion against private expenditure on pesticides of some $4 billion. Of the $8 billion external costs, the authors recognise that $3 billion is paid, effectively, by farmers (pollination losses etc.) leaving a $5 billion bill to be absorbed by society.[14] They estimate benefits associated with reduced crop losses from using pesticides of some $16 billion.

In another US study by Steiner et al (1995), 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.

A Report 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 (see Dinham 1998).

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 (Pearce and Tinch 1998). Globally, this study suggests a figure of $4 billion, apparently on the basis

[13] These include Pimentel et al (1993) and Steiner et al (1995). A study by Fleischer and Waibel has also undertaken a similar exercise in the German context (see Dinham 1998), whilst another study (James 1995) 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).

[14] 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).

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.[15]

A recent study by Pretty et al (2000), 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. Another study by Foster et al (1998) used a contingent valuation survey to estimate estimate an externality per kg of active ingredient of £12 per kg on average. 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).

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 can 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 (Jaenicke 1998). 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 (Meier-Ploeger et al 1996). 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, which translates to (approximately) €13-€19 per kilogramme of active ingredient used. Table 46 below shows that, on a per hectare basis, for 1993-1995 purchases varied between between 1.2-16.3 kilograms of active ingredient used per hectare of arable and horticultural land (including set aside). Assuming, therefore, that pesticide use could be reduced by 20% (and clearly this

[15] 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.).

would vary according to the country concerned, the crops being grown and, therefore, the pesticdes currently in use), 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.

Table 46: Estimated External Benefits Associated With Reductions In Pesticide Use

Pesticide reduction Assumed			**20%**			
Unit externality (low, € per kg a.I.)			***12.8***			
Unit externality (high, € per kg a.I.)			***19.2***			
Country	**Arable and horticultural land incl. Set aside (1000 ha)**	**Avge. crop value 1992-1994 (million €)**	**Avge. quantity of pesticide purchased 1993-1995 (tons a.i.)**	**kg a.i. per hectare**	**External benefit per hectare (low, €)**	**External benefit per hectare (high, €)**
Austria	918	1481	3669	4.00	10.2	15.3
Belgium	747	2600	10282	13.76	35.2	52.9
Denmark	2460	1921	4277	1.74	4.5	6.7
Finland	999	1516	1180	1.18	3.0	4.5
France	15865	22061	88492	5.58	14.3	21.4
Germany	11359	12283	29350	2.58	6.6	9.9
Greece	2111	5914	9260	4.39	11.2	16.8
Ireland	155	532	2523	16.28	41.7	62.5
Italy	8464	20969	78394	9.26	23.7	35.6
Luxembourg	58	38	253	4.36	11.2	16.8
Netherlands	839	7224	11284	13.45	34.4	51.6
Portugal	1578	1362	9426	5.97	15.3	22.9
Spain	12888	13099	29501	2.29	5.9	8.8
Sweden	1394	739	1621	1.16	3.0	4.5
U.K.	5186	6722	33240	6.41	16.4	24.6

Source: Author's caculations using pesticide use data from Wossink and Feitshans (1999).

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 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.

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 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);

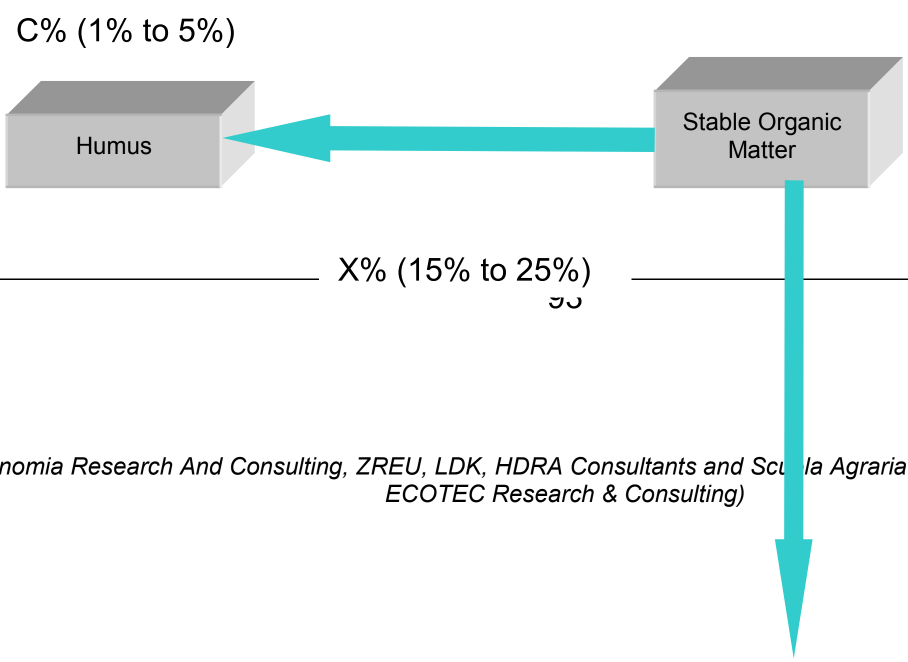
2. The slow or decomposable soil fraction (turnover time two to three years – of great importance to developing good soil structure – disturbed by cultivation and other disturbances – provides a source of carbon for biological digestion by microbes, so linking to the active pool. – can be viewed as mature compost); and

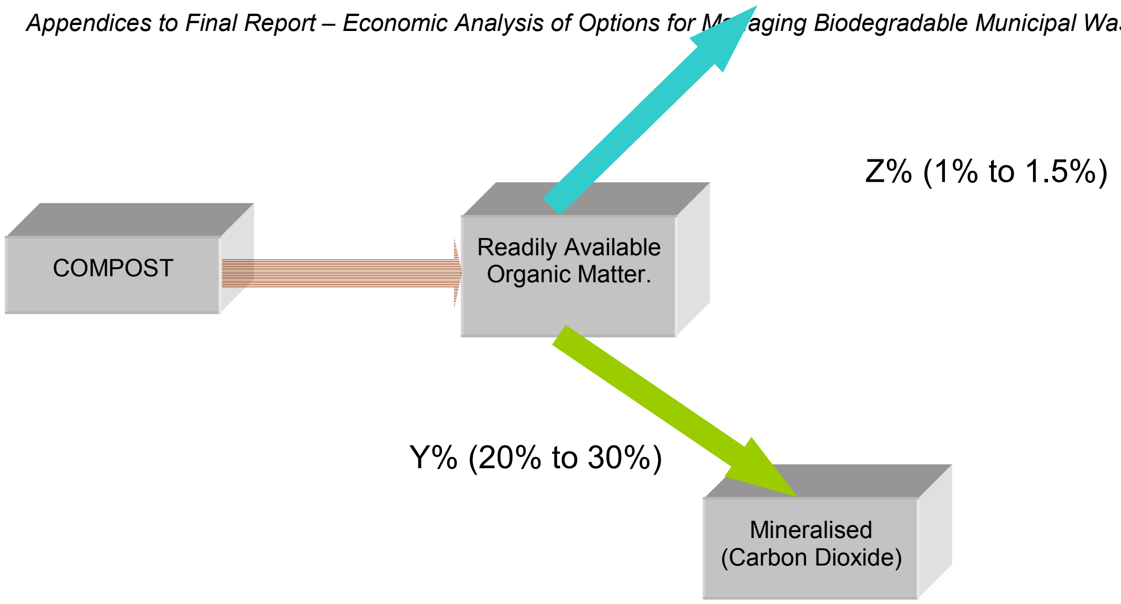
3. The passive soil organic fraction. (turnover time of order 1000 years - resistant to oxidation processes – acts as a ‘cement’ that binds particles).

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 Figure 3 below.

Figure 3: Basic Description Of Modelling Of Fate Of Carbon In Compost / Soil





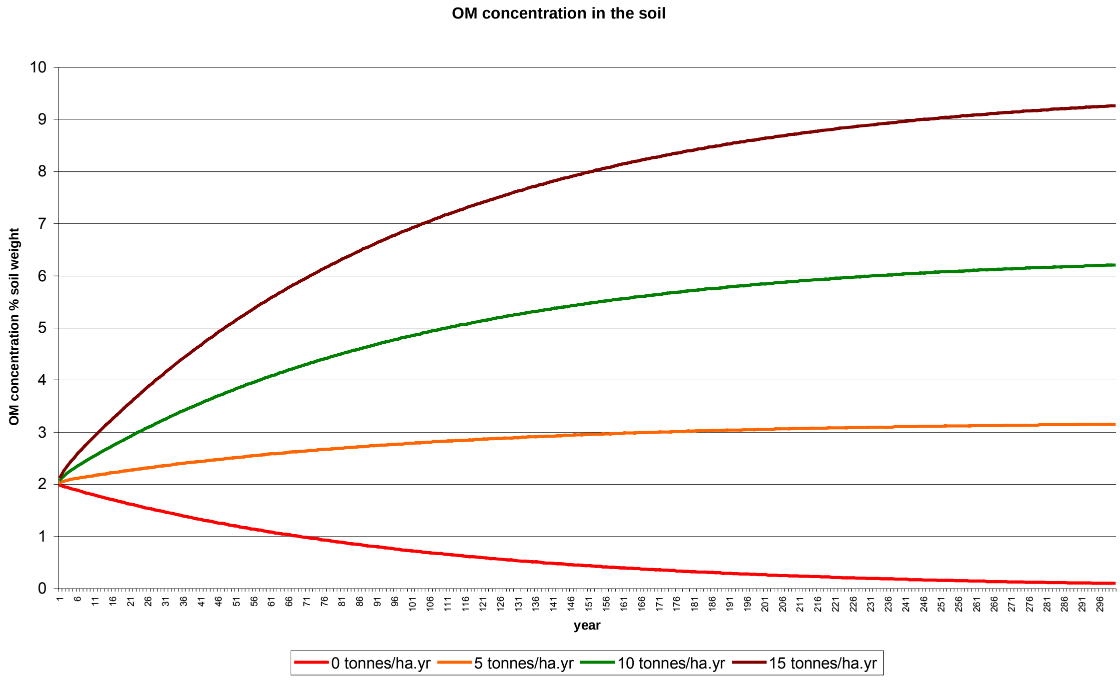
The application of compost is assumed to lead to the readily available carbon being mineralised at y% whilst x% of the readily available organic carbon is converted to stable organic matter. Of this stable organic matter, some carbon is mineralised, but at a much lower rate than that at which the readily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

Using the following figures:

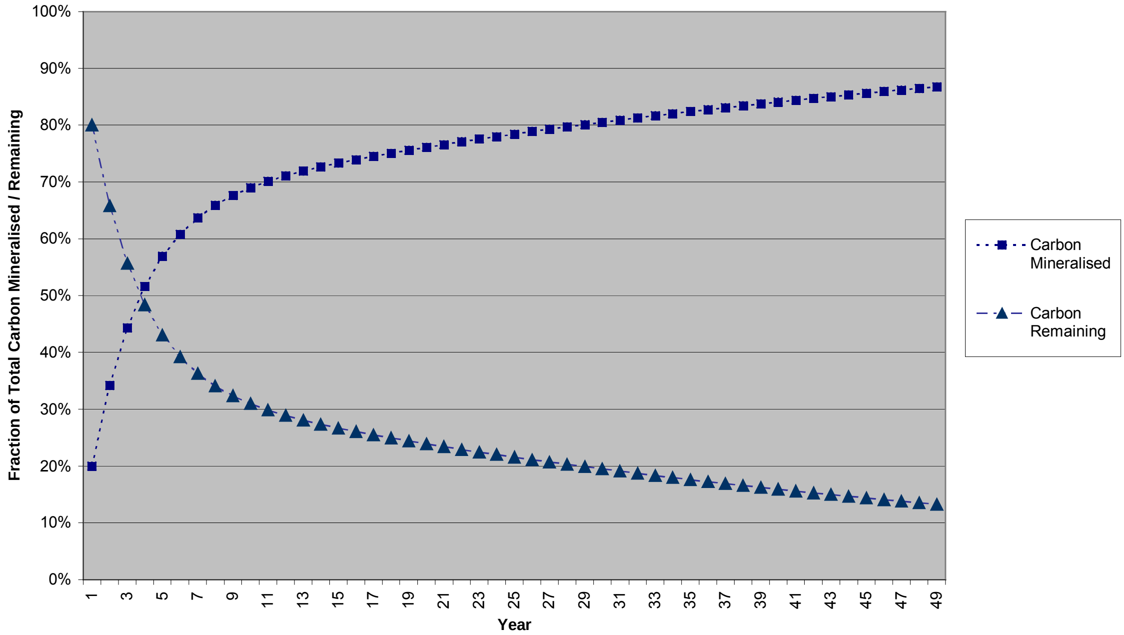
X = 25%, Y = 20%, Z = 1%, and with an initial organic matter concentration of 2%, one can understand the effects of different rates of compost application. This is shown in Figure 4. The production of compost and incorporation in topsoil has the potential to act as a significant reservoir for carbon. When combined with responsible agricultural practices it could have a positive impact on reducing the rate of global warming.

Figure 4: 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 Figure 5 for a 50 year period (after which some 13% of the original soil organic carbon remains in the soil).

Figure 5: Carbon Remaining in Soil / Mineralised Over Time



In our modelling, we have used different parameter values reflecting, principally, temperature differences in the different countries. For the mineralisation rate, Y, we have used the same figures as for the mineral nutrients as discussed above. For the other parameters, the figures are split between Northern and Southern Member States. 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.[16]

[16] Personal comm.. F. Tittarelli.

Table 47 Parameters Governing Transformation of Organic Matter in Compost When Applied to Soil (by Member State)

Parameter	North	South
X	25	15
Y	15	30
Z	1	1.5

These figures generate profiles for carbon dioxide emissions. The external costs from these emissions, discounted over time, are then assessed and added to the model.

Compost as an Alternative to Peat

The area of peatlands represents approximately 5%-8% of the worlds 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. Finland has the highest proportion of pe atlands or bogs, with 10,000,000 hectares, with Ireland and the UK having 1,178,798 ha and 1,341,841 ha respectively, however these are declining rapidly.

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

<u>*Primary Uses of Peat*</u>

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 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 consumers 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 preferred in 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 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 diluting peat with compost or other peat alternatives. In the UK alone, if 10% of all peat usage were to be replaced with an alternative, peat use would fall to 2.8 million m^3 a year as opposed to 3.16 million m^3. As consumer confidence increases in the peat/compost dilutant, so could the levels of dilution.

<u>Environmental Costs of Peat Extraction</u>

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 loss of carbon reservoirs.

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 site 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.

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 in the UK translates values for the Somerset Levels into a value of £7,245 per hectare (Willis et al 1993).[17] Another UK study (Hanley and Craig 1991) estimated a preservation value of £68.4 million, or £4.1 million per annum using a 6% discount rate. Other valuation studies looking at wetland areas are shown in Table 48.

[17] This is the interpretation of the original study from Ingham (1996).

Table 48 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: • average for all types of wetlands • average for flood control • average for biodiversity • average for USA • average for the UK • average for the rest of Europe	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: • use values • non-use values of local population • non-use values of the rest of the UK	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: CVM	WTP for the preservation of the current state of the wetlands: • total use value • total non-use value	 66 / ha 1223 / ha	the estimates are additive. No adjustment factor is available. no. of ha affected no. of ha affected.
Haneman n et al (1991)	USA, San Joachin Valley, California	WTP for • maintenance • improvement	 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 • present condition • some improvement towards natural conditions • recovery to full river condition	 2.8 / user / visit 2.2 / resident / visit 4.0 / user / visit 3.4 / resident / visit 5.0 / user / visit 4.8 / resident / visit	METHOD NOT RECOMMENDED choose which change applies no. of visits by non-residents no. of visits by residents

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).[18] 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 (Roulet et al 1993; Regina et al 1998; Freeman et al 1993). 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.[19]

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.

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.

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[18] 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).

[19] See, for example, Environment Canada (1998).

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 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.[20]

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.[21] 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 Table 49 below.

Table 49 Gaseous Emissions From Peat Draining And Extraction (tonnes per tonne peat extracted)

Pollutant	SO_2	NO_x	TSP	CO_2	CH_4	N_2O
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/m3. The density of compost, on the other hand, is of the order 500kg/m3 for a 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 peatbogs is reduced. This is due to the fact that the non-use values of peatbogs appear to be significant.

Reduction in Pesticide Use

[20] Chapter 12, Ireland, in European Commission (1999a).

[21] Chapter 9, Finland, in European Commission (1999a).

In the United States the Nursery industry uses disease suppressive compost widely and as a consequence have significantly reduced the need to use fungicide drenches, and in some instances replacing the need to use methyl bromide completely (used primarily as a soil fumigant). This is particularly relevant as methyl bromide has been implicated in ozone depletion and consequently the use of methyl bromide is to be phased out globally by 2005.

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 1200 ml pots for the ornamentals 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.

UNQUANTIFIED COSTS AND BENEFITS

Benefits

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 (Schnitzer, 1982) 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 (Jenkins, 1991) have shown an improved efficiency of mineral fertilisers where soil organic matter is high and this has certainly also been seen in other trials. 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 be 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.

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 volatise 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_2O.

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 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.[22]

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 Table 40. 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 Table 50.

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.

[22] Jackson, L. , personal communication, based on series of reports, Researching the Use of Compost in Agricultiure 1997-2001, HDRA Consultants.

Table 50: 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 (Pretty et al 2000). 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 (Hanley 1990). Grossing up on the basis of 835,212 households,[23] the aggregate 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 by Gren (1995), 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.

None of these estimates provides a reliable basis upon which to value the 'displaced leaching' associated with the nitrogen displacement. However, the effect should not be ignored.

[23] 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.

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.[24]

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 (Favoino and Centemero, 1995). 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.

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.

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[24] Jackson, L. personal communication, based on series of reports, *Researching the Use of Compost in Agriculture 1997-2001*, HDRA Consultants.

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 waterways alone were estimated at \$2-\$17 billion NRC 1989; Ribaudo 1989; Pimentel et al 1995). 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 (Evans 1996).

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.

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

Improved Tilth

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.

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 innoculate to the contaminated soil, providing the microorganisms 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.

Micronutrients 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.

Negative Effects 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.

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 Table 51).

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 51: 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) and COWI (2000)

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 odourous 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 maintainance of treatment technologies (e.g. watering the biofilter, preventing its compaction, etc.)

- the release of odourous areas from tipping gates, above all where deep bunkers are used.

A good number of facilities across Europe are curently 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.

Odour Treatment Using Biofilters

Biofilters are used primarily to treat odours.[25] They are an air pollution control technology that use a biologically active, solid media bed to absorb/adsorb compounds from the air and retain them for their 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 circuitation of air (this problem is one of major maintainance 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.[26] The desired qualities of a biofilter are outlined in Table 52.

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

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[25] 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. 116

[26] Evidently, some of these such as the lime and carbon examples, work not through biological activity but through absorption.

the specific loading rate, expressend in $cm_{air}/hr.cm_{biofilter}$). 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 H_2S (>99%), methyl mercaptan, dimethyl disulphide, dimethyl sulphide (>90%) and various terpenes (>98%) (see, e.g. Hartenstein and Allen 1986).

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 maintainance in the short run, while the control of porosity and of the even distribution of inlet air is the main maintainance issue in the medium to long term.

Table 52: 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_3 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 $m^3/h\text{-}m^3$, 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 $gm^{-3}h^{-1}$ *)
Gas distribution	The manifold must be properly designed to present a uniform gas flow to the media

Source: Adapted from Haug (1993) and Swanson and Loehr (1997)

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.

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 high temperatures (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, Alveolitus, Mucus membrane irritation, Chronic Bronchitis and coughs, Gastro-intestinal disorders and Skin disorders.

Individuals who work at a composting facility – less frequently those who are located in close proximity to a facility - particularly when compost is being moved or agitated, may be exposed to, and inhale large quantities of, bioaerosols. To most individuals, exposure to bioaerosols does not cause a problem, however certain individuals for example, asthmatics and the immuno-compromised may suffer adverse health effects after exposure to bioaerosols.

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 (Gilbert 1998). 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 microorganisms;

- 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 53 Presence of Micro-organisms at Compost Site

Microorganisms Species	Shredding cfm/m^3	Turning cfm/m^3	Screening cfm/m^3
Fungi	0.000004	0.0000001	0.000005
Aspergillus fumigatus	0.000001	0.0000001	0.000001
Total bacteria	0.0000001	0.0000001	0.000005
Streptococci	0.00005	0.0006	0.0001
Enterrobacteria	0.00001	0.0001	0.001
Total Actinomycetes	0.000001	0.0000001	0.000005

Source: Environment Agency (2000).

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.

Micro-organisms that are introduced with the feedstock include a range of enteric pathogens, bacteria, viruses and parasites. These are usually present in waste that includes faecal matter, sanitary tissue or, possibly, food. Although pathogens are present in higher numbers in certain waste streams, such as, sewage sludge, it is possible to find contamination in composts made from 'cleaner' materials, e.g. green waste. Enteric micro-organisms are rarely inhaled (Déportes *et al*, 1995). These are therefore of particular importance where it is possible that the compost could be ingested (through poor hygienic practice during and after handling).

There is also the question of risk once the material has been allowed into the environment. Some work has been done to determine persistence and viability of pathogens in the soil. Parasites may remain in soil for some time. Déportes *et al*, (1995) reported that *Ascaris* (round worm) eggs had been found still living up to 107 days after inoculation. Toxacara can last more than 5 years (Déportes *et al*, 1995). Mackenzie (1998) reported that there was no proof that *Escherichia coli* strain 0157, a potentially deadly bacteria, dies in soil within the waiting times advised in the regulations, but pointed out that it can survive in manure applied to the field for months.

The best way to control potential hazards of this kind would therefore appear to be to ensure that disease-causing organisms are not contained in the compost feedstock. The risk factor can be reduced by sorting the raw materials when they arrive at the composting site. Déportes *et al* (1995), in a review of literature, suggests that the microbial hazard from faecal matter was modest. Eliminating faecal matter should lead to a clean safe product. Refusing to accept pet or kennel waste, nappies or tissues from hospitals, for example, would reduce the incidence of disease-forming organisms at the start (Déportes *et al*, 1995).

The process of composting itself can also help to sanitise the final material through both pasteurisation (see Table 54 for the time/temperature combinations suited to ensure sanitation) and the loss of starting hospitable biochemical features.

It should be noted that organisms pathogenic to humans and animals are not in their natural environment when applied to the soil and they would have to compete with others that are (Déportes *et al*, 1995). Ones that can change their state, e.g. by forming a coating to protect them from the hostile environment, may have a chance of survival. These include the spore-forming bacteria that are also difficult to inactivate by pasteurisation.

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 (Johansson et al 1997):

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

Table 54 Temperature/Time Regimes Known To Be Effective During Composting Against A Range Of Pathogens Of Importance To Humans [27]

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

[27] CEN/TC 223 WG2 (1995)

of users against inherent pathogens. Provided all precautions are taken, the risk to humans or animals is believed to be minimal.

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 Table 55, 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 (Johansson et al 1997). 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 anthybiosis (see also section 6.0). Such features are increasingly being exploited also on a commercial level.

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.

Table 55 Inactivation Regimes For A Range Of Pathogens Of Importance To Growers

Pathogen	Inactivation regime
Fungus with resting spores	
Phytophthora infestans(potato blight)	2 - 3 wks: 47 - 65 °C
Phytophthora cryptogea	2 - 3 wks: 64 - 70 °C
Fusarium oxysporum	30 minutes: 57.5 - 65 °C
Fungus with sclerotia	
Sclerotinia sclerotiorium (white mould)	2 - 3 wks: 64 - 70 °C
Bacteria	
Pseudomonas syringae pv phaseolicola (halo blight of bean)	4 days: 35 °C
Erwinia amylovora (fire blight of pome fruits)	7 days: 40 °C under optimum conditions
Viruses	
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
Nematodes	
Heterodera rostochiensis	50 - 55 °C
Aphelenchoided fragarie	1 hr: 50 °C/4 hrs - 44 °C
Ditylenchus dipsaci	1 hr: 50 °C/4 hrs - 44 °C
Meloidogyne hapla	1 hr: 50 °C/4 hrs - 44 °C

Source: Johansson et al (1997).

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 facilites located at landfill sites. Smaller birds are occaisionally 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

It is very difficult to find a location where rats are not present and a composting facility, especially if situated in a rural location is no exception. Rats are 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; most often it is necessary to emply professional rodent control operatives to prevent populations getting out of control.

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.

Heavy Metal Concentrations

Much 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 (Baker and Amacher 1982). 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 heavy metals by plants or the consequences of heavy metals once in the food chain (perhaps unsurprisingly given the range of issues affecting uptake).

Poletschny (1992) studied 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. After a 10 to 15 year period of application Poletschny 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 (Tisdell and Breslin 1995). 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 Table 56).

Table 56 Cadmium Contents Of Main Commercial Phosphate Rocks According To Different Sources.

Origin	**Cadmium content (mg per kg P_2O_5)**	
	(1)	**(2)**
Igneous		
Kola (Russia)	< 13	0.25
Pharlaborwa (South Africa)	< 13	0.38
Sedimentary		
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
Youssoufia (Morocco)	121	120
Gafsa (Tunisia)	137	173
Togo	162	147
North Carolina (USA)	166	120
Taiba (Senegal)	203	221
Nauru	243	

Sources: (1) Davister (1996). (2) Demandt (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.

Trace Elements and Compost[28]

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 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 organisims 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.

<u>Elements Of Concern Primarily To Plant Health</u>

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.[29]

<u>Elements Of Primary Concern To Animal And Human Health</u>

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. Converseley, 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

[28] See Woodbury (1998).

[29] Leachate should be captured and treated and not allowed to drain into soil/watercourses.

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.

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.

RESULTS OF BENEFITS ANALYSIS

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

Results of recent work concerning the market shares for compost made from source-separated materials are given in Table 57 and Table 58 below. The relative shares reflect, in part, the nature of the materials produced, but there is also some effect from the nature of agriculture in the country concerned. The large amount of nutrients from farm wastes in, for example, Denmark, may reduce the demand for compost as a nutrient substitute since animal manures are already quite widely used. In the Netherlands, when the MINAS system of accounting for mineral surpluses was introduced, the demand for compost fell, reflecting the more efficient application of manure to the land.

Table 57 Market Shares (%) Of Compost Sales In Selected Countries

	Austria (1998)	Belgium (1998)	Germany (1998)	Denmark (1998)	Italy (1999)	Netherlands (1998)	Market size
Landscaping	30	24	25	19	30	30	Large
Landfill Restoration	5	5		13		-	Small
Agriculture + Special cultures	35[1)]	5	43	10	20	40	Very big
Horticulture	5	6	5	3			Medium
Earth works	5	33	10	-	50		Medium
Hobby gardening	20	18	14	48		20	Large
Export		9	-	-		-	Very small
Miscellaneous		-	3	7		10	

Source: Barth (2000)

Table 58 Size Of Application Areas For Compost In Some European Countries/Regions (% of total used/marketed compost, average of 1996-98)

	Denmark	Flanders, B	Germany	Austria
Private gardens	49	18	16	20
Park & landscaping	19	27	37	30
soil mixing companies	-	11	-	
other wholesalers	-	8	-	
Horticulture	2	6	12	10
growth media/potting soil	-	14	-	
Agriculture	10	5	32	30
Landfill reclamation/soil sanitation	15	4	-	5
Export (agriculture/wine)	-	7	-	
Others	5	-	3	5
No. of people in the area (mill.)	*6*	*5*	*80*	*8*
Total amount of used/marketed compost in 1998 (1.000 tonnes)	*280*	*201*	*4,100*	*300*
Composition of raw material (% w/w of all input-material ending as compost, 1996-98)				
Municipal biodegradable waste	*7*	*21*	*41*	*51*
Yard waste	*88*	*79*	*59*	*26*
Others (often dewatered sewage sludge)	*5*	*-*	*-*	*23*

Source: Amlinger (2000)

These figures relate principally to compost produced at specific plants. In some countries, home composting and (notably, in Austria) on-farm composting play an important role, whilst community composting schemes also can play an important role, especially in sparsely populated small communities. Amlinger (2000) writes:

… home or backyard composting plays an important role in some countries. The composting of materials on the private property (backyard and enterprises), that are not collected and treated in composting facilities in Austria is estimated as 40 % (w/w) of the total potential of organic waste. In the province of Lower Austria it was calculated that a 60 % (w/w) of the total figure of organic materials is treated as backyard composting (biowaste and yard waste) which is related to 56 % of the households.

To this end, the actual amount of material being used in agricultural and hobby-gardening applications is probably rather higher than is reflected in the statistics above, with the role of the other markets correspondingly reduced once home and community composting are taken into consideration.

Consequently, we have assumed that, for every tonne of waste composted, 45% is used in agricultural applications, 35% is used in hobby gardening, 5% is used in horticulture and 15% is used in landscaping and landfill restoration. For the landfill restoration and landscaping aspects, we assume that there are no external benefits derived from use (though there may well be some associated with, for example, enhanced biological activity in the soil, encouraging plant growth and reducing run-off rates). In practice, the end-use markets for the use of compost will be different in different countries. For example, in those countries where there is widespread use of manure in agriculture, the degree to which compost penetrates the agricultural market may be more limited (although evidence of the benefits of compost use in grasslands exists). This to some extent represents the case in Flanders, where VLACO has worked hard to diversify outlets for compost outside agriculture.

As well as finding different outlets in different Member States, compost utilisation is likely to 'displace' different products in different countries. The use of peat-based soil improvers in home gardens is not universal practice, and indeed, the market for soil improvers is increasingly diverse. Our assumptions concerning product displacement effects are as shown in Table 59. The double column, 'unit effects', gives the unit externality per tonne of waste composted associated with different external benefits. The marketing weights are an estimate of the fraction of each tone of compost which could be expected to 'generate' the associated benefits. These are clearly an over-simplification of what would most likely be a much more complex reality, with variation from one Member State to another both in the quantities marketed for specific markets and the products which are effectively being displaced in those markets.

The double column 'replacement rates' then gives low and high rates at which these benefits could actually be attributed to the compost as applied. This depends upon the degree to which users reduce their use of avoided products. So, for example, it is suggested that for each tonne of compost used, between 40% and 100% of the fertiliser use which can be displaced by use of compost actually is displaced. This effectively assumes that farmers do not, in general, re-optimise fertilisation perfectly. On the other hand, they may well come quite close the more they understand the product being used.

Table 59: Marketing Rates and Replacement Rates Used Assumed in Benefits Assessment

	Unit Effects (€/tonne)		Marketing Weight	Replacement Rate	
	Low	High		Low	High
External Benefits from Nutrient Displacement	0.83	4.36	0.40	0.40	1.00
External Benefits from Pesticide Reduction	0.50	0.75	0.80	1.00	1.00
External Benefits from Avoided Nitrous Oxide Emissions	0.24	1.62	0.40	0.40	1.00
External Benefits from Avoided Process Wastewater Disposal	0.08	0.08	0.40	0.40	1.00
External Benefits from Avoided Peat Extraction	0.89	1.31	0.40	1.00	1.00

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, th 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.

Private Savings from Avoided Fertiliser Use

A recent European Commission report (Oosterhuis et al 2000) suggested a market price of phosphoric acid around USD 400 per tonne P_2O_5, and the price of phosphate fertilisers (superphosphate 44-46%; diammonium phosphate) between USD 250 and 300 per tonne of product. This implies a figure of the order €500-550 per tonne P_2O_5. This fits reasonably well with our own investigations which suggest a cost pf €0.47 per kg P_2O_5. We have used the figure €0.5 per kg P_2O_5.

For nitrogen, the price varies quite considerably depending upon the form in which the nitrogen is purchased. For example, we obtained quotes for €0.19 per kg for urea with 46% N content, implying a cost of €0.41 per kg N. Ammonium nitrate, with 26% N, was quoted at €0.17 per kg, or €0.65 per kg N. We have used a figure €0.53 per kg N. We have used a figure of €0.6 per kg on the basis of a quote for potassium sulphate, with 50% K_2O, of €0.3 per kg.

Under the fertilizer replacement scenarios outlined above, this would lead to net savings of €4.7 per tonne of municipal waste composted. This is similar to the external costs of composting a tonne of municipal waste.

External Costs and Benefits of Composting

The results of the analysis are shown for different discount rates for the case of Austria. These illustrate that the total analysis is dominated by the external costs of greenhouse gas emissions, with other emissions and the positive benefits playing a minor role. For this reason, as the discount rate increases, so the external costs fall also.

The benefits side suggests that only at the higher discount rates do they approach the magnitude of the external costs. These also vary with the discount rate chosen but less so than the external costs from greenhouse gases. Because of these factors, the analysis varies relatively little across countries.

It has to be recalled that the analysis is incomplete. External costs which have not been captured have been outlined above, as have the unquantified external benefits. Even those benefits which have been captured are subject to a high degree of uncertainty. It should be recalled that this is one of the first attempts to capture these benefits in anything approaching a meaningful way.

Table 60: External Costs and Benefits of Composting, Austria, 1% Discount Rate

Discount Rate 1%		**AUSTRIA**	
		Low	**High**
Greenhouse Gases			
	Carbon Dioxide (process)	-17.97	-18.79
	Carbon Dioxide (post application)	-5.50	-5.75
	Methane	-0.46	-0.61
	Nitrous Oxide	-0.21	-0.32
Other Air Emissions		-0.02	-0.08
Energy Use (kWh)	**50**	**-0.62**	**-1.47**
Fuel Emissions (litres)	**1**	-0.66	-1.03
Total External Costs		-25.45	-28.06
External Benefits from Nutrient Displacement		0.25	2.74
External Benefits from Pesticide Reduction		0.17	0.26
External Benefits from avoided nitrous oxide emissions		0.08	1.29
External Benefits from avoided process wastewater disposal		**0.01**	**0.03**
External Benefits from avoided peat extraction		**0.75**	**0.91**
Net Externality		-24.18	-22.82
Net Externality (no CO2)		-6.21	-4.03
Memorandum Items			
Private savings from avoided fertiliser use		0.76	1.91
Private savings from avoided pesticide use		0.43	0.43

Table 61: External Costs and Benefits of Composting, Austria, 3% Discount Rate

Discount Rate 3%		**AUSTRIA**	
		Low	**High**
Greenhouse Gases			
	Carbon Dioxide (process)	-8.56	-9.01
	Carbon Dioxide (post application)	-2.30	-2.42
	Methane	-0.44	-0.48
	Nitrous Oxide	-0.09	-0.16
Other Air Emissions		**-0.02**	**-0.08**
Energy Use (kWh)	**50**	-0.62	-1.47
Fuel Emissions (litres)	**1**	-0.57	-0.95
Total External Costs		-12.61	-14.57
External Benefits from Nutrient Displacement		0.13	1.66
External Benefits from Pesticide Reduction		0.17	0.26
External Benefits from avoided nitrous oxide emissions		0.03	0.58
External Benefits from avoided process wastewater disposal		0.01	0.03
External Benefits from avoided peat extraction		0.43	0.57
Net Externality		-11.84	-11.46
Memorandum Items			
Private savings from avoided fertiliser use		0.76	1.91
Private savings from avoided pesticide use		0.43	0.43

Table 62: External Costs and Benefits of Composting, Austria, 5% Discount Rate

Discount Rate 5%		**AUSTRIA**	
		Low	**High**
Greenhouse Gases			
	Carbon Dioxide (process)	-2.34	-3.38
	Carbon Dioxide (post application)	-0.57	-0.82
	Methane	-0.16	-0.23
	Nitrous Oxide	-0.02	-0.03
Other Air Emissions		-0.02	-0.08
Energy Use (kWh)	**50**	-0.62	-1.47
Fuel Emissions (litres)	**1**	-0.52	-0.90
Total External Costs		-4.25	-6.91
External Benefits from Nutrient Displacement		0.07	0.97
External Benefits from Pesticide Reduction		0.17	0.26
External Benefits from avoided nitrous oxide emissions		0.01	0.12
External Benefits from avoided process wastewater disposal		0.01	0.03
External Benefits from avoided peat extraction		0.23	0.37
Net Externality		-3.77	-5.17
Memorandum Items			
Private savings from avoided fertiliser use		0.76	1.91
Private savings from avoided pesticide use		0.43	0.43

A.5.0 EXTERNAL COSTS OF ANAEROBIC DIGESTION

INTRODUCTION

Conducting life-cycle analyses of the anaerobic digestion process is just as complex as for composting (if not more so due to the added complication, and sensitivity to, the collection of biogas for energy generation, which varies across the different systems used). Some of the issues are similar to those described above for composting, and again there are a number of different process types. The fact that the processes are biological ones, that different process technologies are used at different operating temperatures, and that they work with varying feedstocks makes it difficult to characterise emissions through reference to one set of figures. In situations where one assumes fossil-fuel energy is being displaced, the assumptions concerning net energy supplied are again important.

Possibly because AD technology is comparatively new, and also because the process is less widespread at present, there appears to be less by way of emissions data available compared with composting. The data sources available are examined below.

The environmental effects associated with AD are numerous and have to be considered alongside any environmental impacts end-product use might be reducing. Therefore this section will be broken down into an examination of the AD process itself, including the various variables that exist and how these impact relatively on the environment; the environmental impacts minimised through end-product use; and a brief comparison with other treatment options including composting. Finally, a brief discussion on which of the environmental impacts discussed can be quantified and valued reliably in the model.

Process Characteristics

There are a number of variables associated with the AD process which will affect the environmental impacts.

- **Temperature**: As stated above, practical AD systems are operated under either mesophilic (20-45^{o}C but usually 35^{o}C) or thermophilic (50-65^{o}C but usually 55^{o}C) conditions. The reactor temperature must be maintained at a relatively constant level to maintain the gas production rate. Thermophilic digestion generally leads to a higher gas production rate. This will increase the amount of recoverable energy, but this is offset by the increased energy requirement associated with thermophilic digestion which requires additional heating (which mesophilic generally does not). The higher temperature process also reduces the retention time necessary (see below).

- **Retention time**: A higher retention time will obviously enable more extensive biodegradation and subsequently a better quality digestate and smaller environmental and health impacts on application. This has to be balanced against a lower possible loading rate, reducing the throughput and thus increasing the economic cost per tonne treated (see below).

- **Waste type**: As mentioned above, the feedstock characteristics have very important effects on the AD process. A high quality feedstock will increase the quality of the digestate and minimise the concentration of potentially toxic materials, minimising subsequent environmental impacts associated with application. Moisture content is also important in terms of a low value increasing both ammonium inhibition of the AD process, and salt toxicity. High heavy metal concentrations can also be toxic to methanogenic bacteria in the following order (of increasing severity): iron < cadmium < zinc < chromium < lead < copper < nickel. The volatile solids content will affect the extent to which the process needs to be monitored to avoid the damaging effect of over-loading.

GASEOUS EMISSIONS DATA

Some gaseous emissions data from plant themselves represents the biogas as it is produced from the process. Table 63 gives manufacturers' data for the DRANCO plant in Salzburg. Yet energy is derived from combustion of gases. The combustion process and air pollution control technology will, together, determine the ultimate emissions to air. The combustion process may occur at a separate facility in which the delivery of gas to that facility might incur some small losses.

For the purposes of this work, we are more interested in the gaseous emissions ultimately arising after the combustion of biogas.

White et al. estimate the gaseous emissions from anaerobic digestion. CO_2 emissions are estimated to be 440,000 g per tonne of waste. There are a number of gaseous emissions that would be expected from the combustion process. These include heavy metals, dioxins, NOx and SOx but the emission levels of some of these species is very low. The data does not quote emissions of CH_4 or N_2O. One might expect near-zero emissions of methane if the combustion process was highly efficient. 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 such a process is unlikely to result in combustion of 100% of the methane component.

Furthermore, combustion of the gas seems likely to lead to some emissions of N_2O. Therefore it is likely that White et al. underestimate the emissions of certain greenhouse gases. As in the case of the compost aspect described above, the IWM2 model also seems to have the same error in terms of a missing factor of 10,000.

The only other complete data source this study was able to use with respect to gaseous emissions from the anaerobic digestion process was the ORWARE model as described above for the composting process. In a similar way, the model calculates the emissions depending on the composition of the feedstock waste. In this case however the calculation produces the levels of biogas production expected depending on the waste composition. This derived volume is important in both calculating the final emissions to air after combustion (for which the model has factors relating gaseous emissions such as CO_2 to MJ of methane produced), as well as calculating the amount of energy that can be generated through combustion.

Table 63 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_2	Vol %	31	35.2
O_2	Vol %	1.1	0.9
N_2	Vol %	4.1	3.9
CO	µg/m^3	nd	Nd
H_2S	mg/m^3	284	289
1,1,1,-Trichloroethane	µg/m^3	nd	Nd
Trichloroethene	µg/m^3	nd	nd
Tetrachloroethene	µg/m^3	nd	nd
vinyl chloride	µg/m^3	nd	nd
1,1-dichloroethene	µg/m^3	nd	nd
dichloromethane	µg/m^3	nd	nd
chloroform	µg/m^3	2	nd
1,1-dichloroethane	µg/m^3	nd	nd
1,2-dichloroethane	µg/m^3	nd	nd
benzene a	µg/m^3	70	50
toluene a	µg/m^3	220	250
Ethylbenzene	µg/m^3	610	630

M+p+o xylene	µg/m^3	360	290
total Chlorine	µg/m^3	1.5	nd
total Fluorine	µg/m^3	nd	nd

Source: Environment Agency (2000)

This study utilised the model to calculate the volume of biogas produced when source separated organic waste is digested. The results ranged from around 310 m^3 to 364 m^3 biogas per tonne of waste (depending on the composition of the biogas). This estimation is relatively high compared to the Environment Agency study which predicts biogas generation to be of the order of 100 m^3 per tonne. A study by NOVEM suggested ranges from 75-140 m^3.

The composition of waste used in the model was the same as that used in the composting calculation above. In reality however, these compositions may not be the same because different feedstocks may be used, therefore affecting the carbon and volatile solids content. The compositions are also given in dry matter, which needs to be converted to wet matter to calculate emissions per tonne of waste. The same dry matter content was used for both composting and AD calculations.

Using a range of biogas compositions (with variations of CH_4 and CO_2) along with the factors of gaseous emissions per MJ of methane provided in the ORWARE model (see Table 64 below), this study calculated the final emissions per tonne of source separated organic waste digested. The model resulted in predictions for CO_2 emissions ranging between around 350,000 and 520,000 g per tonne of waste. These are relatively close to the estimations in White et al. outlined above. However related estimates for emissions of gases such as SO_x and $N_2O$are very different (for SO_x, around 1,000 g from ORWARE compared to 2.5 g per tonne from White et al.).

The modelling of external costs is very sensitive to these assumptions about SO_x and N_2O (especially, for the latter, at low discount rates). For example, for each of the changes suggested, using the high unit damage costs, the externalities change by around €10. hence, there is a suggestion that the data concerning AD emissions need to be improved considerably in order for us to have confidence in their use. Note that the ORWARE model for emissions of non-greenhouse gases are derived from rather dated factors relating to the use of a gas engine. The emissions of the gases might be expected to be considerably lower in the presence of suitable pollution control equipment.

Table 64: ORWARE Data Concerning Gaseous Emissions from Anaerobic Digestion

Gas	g per MJ of methane
CO_2	85

CH_4	0.1
CO	0.25
N_2O	0.2
SO_x	0.15

Because of the uncertainties in the calculations (as outlined above), this study has used the emissions from the White et al. study in its externality calculations for gaseous emissions from the AD process (Table 65). These are the same data as were chosen for use by Aumonier (1999) on the basis of an earlier literature review. The comments made above concerning data for N_2O and SO_x should be borne in mind.

Table 65 Emissions Data Used For Anaerobic Digestion (per tonne of waste)

Compost Production (kg)	300
Energy Consumption[a] (electrical)	50 kWh
Energy Production[a]	160 kWh
Air Emissions (g / tonne biowaste)	
CO2	440,000
CH4	0
NOx	10
N2O	0
SOx	2.5
HCl	0.011
HF	0.0021
H2S	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 et al (1995)

Energy Related Emissions Data

Anaerobic digestion is likely to lead to a positive energy balance because the biogas production levels provide sufficient energy to run the process plant and export the remaining electricity generated. This study has examined a number of different estimations of both energy consumption and production in the process. Typical engine efficiencies appear to be of the order 30%.

Utilising the ORWARE model above, one can calculate the amount of energy generation depending on the calculated levels of biogas production as well as the energy content of the biogas. However, because of the uncertainties associated with our use of the model (outlined above), along with the fact that we do not have data for on-site energy consumption in plants covered by the ORWARE model, this study has used alternative data sources for AD energy data.

Biogas is typically combusted in an engine, producing both electricity and heat. Some electrical energy is used internally for running the plant, with the rest available for export. In some, though not all, cases, the heat may be recovered too. Biogas can also be used to power waste collection vehicles (as is done in Sweden). However, these applications are less common and have not been evaluated here.

Despite the range of gas volumes from $75m^3$ to $140m^3$ quoted by NOVEM there is no obvious link to the relative proportion of methane in the gas. This probably reflects different processes and feedstocks.

Unlike the case with emissions described above, there are a number of different sources which quote figures for energy production and consumption associated with the AD process. Some of these sources examined in this study are outlined in Table 66 below. It is not always clear in the studies concerned whether the energy generated is heat and electricity, or only electricity. Some studies, such as the NOVEM study, are clear that this is net electricity generation.

Table 66 Anaerobic Digestion - Net Energy Production Figures From Various Sources (quoted in kWh per tonne of waste)

Study	Net Energy Production	
	Minimum*	Maximum*
White et al (2000)	110	
IEA Bioenergy (1997)	75	150
IWM (1998)	100	200
Waterman BBT (1999)	100	
DHV study	102	
NOVEM	21	154

*If only one figure is quoted, the study in question did not provide a range

Other quoted estimates from a recent study are given in Table 67.

Table 67 Net Energy Production Figures From Specific Technological Ad Processes (in kWh per tonne of waste)

Process	Net Energy Production	
	Minimum*	Maximum*
Dranco	105	157
Kompo	85	90
DBA	45	60
WAASA	120	170
Plaunener-Verfahren	85	110

D.U.T	254	292
AN-Anaerob	38	60
BTA	100	130
Prethane-Biopaq	80	140
Schwarting-UHDE	154	

Source: Cited in Tobin Environmental Services (1999). Note if only one figure is quoted, no range was given.

If one uses ranges for the gas generated and the methane content of gas, estamtes for figures for the energy generated and exported (in both electrical and heat forms) can be derived (see Table 68).

Table 68: Electricity and Heat Generated from Anaerobic Digestion

Parameter	Low Value	High Value
Biogas yield	70 m^3/t waste	140 m^3/t waste
Percentage methane	55%	60%
Calorific value of biogas	385 kWh/t waste	840 kWh/t waste
Electricity generated (30% efficiency)	116 kWh/t waste	252 kWh/t waste
Electricity for export (70% of elec. gen)	81 kWh/t waste	176 kWh/t waste
Heat recovered for CHP option (70%)	189 kWh/t waste	412 kWh/t waste
Heat exported for CHP option (80 % of that recovered)	151 kWh/t waste	329 kWh/t waste

These are broadly in line with figures in the preceding Tables. On the basis of these, we derive a range of 81 to 176 kWh per tonne of waste for the net electricity production from the AD process. The heat energy generated is assumed to range from 151 to 329 kWh. These are wide ranges, and probably reflect not only the differences in performance across plant, but also across input feedstocks. We do not have a basis to link gas generation and composition to waste composition in a sufficiently reliable manner to justify making calculations of such a nature. Note that these figures are net of energy use on-site.

BENEFITS FROM USE OF END-PRODUCTS

Much of the analysis concerning product displacement follows similar lines to the analysis carried out in the composting work. This implies some important assumptions:

- First of all, we assume that the material is stabilised through an aerobic composting phase;
- Second, we assume that all excess water is discharged to sewer rather than being put to any use; and
- It is assumed that the composted material finds the same outlets as those which are assumed for composted biowaste.

Some changes to the analysis are made. Table 69 reports on nutrient values of digestate from different digestion plants.

Generally, the nitrogen content of compost is of the same order as for the compost plant but with the phosphorous and potassium contents slightly lower. We have used dry matter content of 1.5% for N, 0.6% for K_2O and 0.6% for P_2O_5. It Should be noted that the analysis is not enormously sensitive to these changes. The mass of compost produced per tonne of input waste varies across different plant and upon the input wastes to the digester. The Dranco plant generates 220kg compost per tonne of input waste. Others, however, generate quantities which are consistently similar to those produced in standard composting plant. As such, the avoided external costs remain somewhat similar.

We do not repeat the analysis here. The reader is referred to the previous Appendix.

Other Emissions

A major difference between composting and anaerobic digestion is the level and character of leachate produced (in net terms, after the potential for re-circulation is exhausted) in the anaerobic system. The quantity of waste, or excess water generated depends upon anumber of factors. Most processes seek to extract excess water from digestate prior to aerobic composting of the remaining biomass. In some countries (and processes), however, little or no attempt is made to do this and the digestate is used on land as a soil conditioner.

Generally, the amount of excess water depends upon:

- The extent of biodegradation;
- The moisture content of input wastes;
- The extent of re-circulation of process water;

Table 69 Characteristics of AD Plants

Company	Country	Feedstock	Units	Nitrogen	Phosphorous	Potassium	Magnesium	Calcium	Sale price (€/tonne)
Avecon / Citec Int	Finland	Biowaste / RDF	% of TS	1.2	0.68	0.74			0
MAT / BTA Biotech	Germany	Source sep. MSW	Ppm	20.0	11.9	14.7	11.6	49.7	60
Dranco (OWS)	Belgium	Source sep. MSW	% of DM	1.90	0.66	0.63	-	-	6.9
Kompogas / Buhler	Switzerland	Organic fraction MSW	Ppm	1-1.3	6-12	8-12	17-26	60-110	0
Linde	Austria	Source sep. MSW	Ppm						22-27
Paques	Netherlands	Fruit / veg from market	Ppm	21.9	9.5	10.5	4.7	-	0
Steinmuller Valorga	France	Unsorted MSW	Ppm	11	8	10	-	-	0
Steinmuller Valorga	Netherlands	Source sep. MSW	Ppm	11	8	10	-	-	7.5
WMC Resource	UK	Unsorted	Ppm	19	13	15	3.67	-	Nd

Recovery		MSW							

Source: IWM (1998).

- The way in which digestate is used (it may be applied direct to land as slurry in some cases, typically in Denmark and Sweden); and
- The degree to which steam is used to heat biomass.

One study sites a figure of 100kg per tonne of waste (wet weight) (Environment Agency 2000). Yet a recent NOVEM review suggests that this is low, with the range being from 0.1-0.5 m3 per tonne (or 100-500 kg per tonne).

The characteristics of excess water are likely to be more polluted from dry systems since more water is re-circulated in the wet systems. Figures for wet and dry systems are given in Table 70 below.

Table 70: Typical Wastewater Characteristics from Anaerobic Digestion

Components	Dry Systems	Wet Systems
COD (mg O_2/l)	20,000 – 40,000	6,000 – 24,000
COD (mg O_2/l)	5,000 – 10,000	2,500 – 5,000
Total N (mg N/l)	2,000 – 4,000	800 – 1,200

Source: IWM (1998).

These figures relate to wastewater before removal of suspended solids. In the post-treatment phase, the liquor from the digestate may be subjected to processes of denitrification, or filtered and / or fed to a decanter, with solids potentially being added to the digestate and the excess water delivered to sewer. Figures from the Dranco plant after centrifuging are given in Table 71. Figures from White et al are given in Table 72.

From the perspective of quantifying the external costs associated with the generation of excess water, in this analysis, the external costs are assumed to be internalised through the costs of treating the water and / or disposing of excess water to sewer. This is not strictly accurate, and constitutes an approximation to the true situation.

UNQUANTIFIED COSTS AND BENEFITS

There are other problems associated with anaerobic digestion which are also of concern. These are similar to those described under the previous Appendix on composting. Odour from the process is an issue, though as with composts, these can be reduced through use of biofilters. Bioaerosols may also present some risk to immuno-compromised individuals. The application of heavy metals to the soil also has to be acknowledged, and hence, as with composts, the desirability of quality source separation is clear. Unfortunately, these external costs cannot be quantified without much more detailed study (and even then, they may be difficult to quantify given the lack of relevant dose response functions).

Table 71: Waste Water Characteristics for DRANCO Plant in Salzburg

Type	Unit	Measurement
Dry Solids	mg / l	17,000 – 27,000
PH		7.5 -8.5
Electrical Conductivity	micro S/cm	15,000 – 21,000
Total N	mg N/l	2,000 – 3,500
NH_4 – N	mg N/l	1,500 – 2,500
NO_2 – N	mg N/l	0 – 10
NO_3 – N	mg N/l	0 – 10
Temp	Centigrade	< 35
Cl^-	mg Cl/l	1,500 – 3,000

Source: Environment Agency (2000).

Table 72: Emissions to Water from Anaerobic Digestion

Water Emissions (g)	
BOD	19
COD	73
Ammonium	29

Source: White et al (1995).

RESULTS

Results for the Austrian case at different discount rates are shown in Table 73, Table 74 and Table 75 below. As with compost, the inter-country variation is not large owing to the limited influence of effects other than global warming. This means, however, that the analysis is sensitive to the discount rate used.

The benefits associated with energy recovery do vary across countries so this leads to some inter-country variation. However, this is not considerable as the main report suggests.

Table 73: External Costs of Anaerobic Digestion for Austria, 1% Discount Rate (€/tonne)

Discount Rate 1%		
	AUSTRIA	
	Low	High
Greenhouse Gases		
Carbon Dioxide (process)	-20.54	-21.48
Carbon Dioxide (post application)	-5.50	-5.75
Methane	0.00	0.00
Nitrous Oxide	0.00	-1.46
Other Air Emissions	-0.13	-0.22
Landfill Leachate		
Avoided External Costs From Energy Generation		
Electricity	1.01	5.18
CHP	1.45	7.44
Total external costs, no displaced burdens	-26.17	-28.91
Total external costs, displaced burdens from electricity	-25.17	-23.72
Total external costs, displaced energy from CHP	-24.73	-21.47
External Benefits from Nutrient Displacement	**0.23**	**2.66**
External Benefits from Pesticide Reduction	**0.17**	**0.26**
External Benefits from avoided nitrous oxide emissions	0.08	1.29
External Benefits from avoided process wastewater disposal	0.01	0.02
External Benefits from avoided peat extraction	0.75	0.91
Total external costs, no displaced burdens	-24.93	-23.76
Total external costs, displaced burdens from electricity	-23.93	-18.58
Total external costs, displaced energy from CHP	-23.49	-16.33
Memorandum Items		
Private savings from avoided fertiliser use	0.57	1.43
Private savings from avoided pesticide use	0.43	0.43

Table 74: External Costs of Anaerobic Digestion for Austria, 3% Discount Rate (€/tonne)

Discount Rate 3%		
	AUSTRIA	
	Low	High
Greenhouse Gases		
Carbon Dioxide (process)	-9.78	-10.30
Carbon Dioxide (post application)	-2.30	-2.42
Methane	0.00	0.00
Nitrous Oxide	0.00	-0.71
Other Air Emissions	**-0.13**	**-0.22**
Landfill Leachate		
Avoided External Costs From Energy Generation		
Electricity	1.01	5.18
CHP	1.45	7.44
Total external costs, no displaced burdens	-12.21	-13.65
Total external costs, displaced burdens from electricity	-11.20	-8.47
Total external costs, displaced energy from CHP	-10.77	-6.21
External Benefits from Nutrient Displacement	0.11	1.59
External Benefits from Pesticide Reduction	0.17	0.26
External Benefits from avoided nitrous oxide emissions	0.03	0.58
External Benefits from avoided process wastewater disposal	0.01	0.02
External Benefits from avoided peat extraction	0.43	0.57
Total external costs, no displaced burdens	-11.46	-10.63
Total external costs, displaced burdens from electricity	-10.45	-5.44
Total external costs, displaced energy from CHP	-10.01	-3.19
Memorandum Items		
Private savings from avoided fertiliser use	0.57	1.43
Private savings from avoided pesticide use	0.43	0.43

Table 75: External Costs of Anaerobic Digestion for Austria, 5% Discount Rate (€/tonne)

Discount Rate 5%		
	AUSTRIA	
	Low	High
Greenhouse Gases		
Carbon Dioxide (process)	-2.67	-3.86
Carbon Dioxide (post application)	-0.57	-0.82
Methane	0.00	0.00
Nitrous Oxide	0.00	-0.15
Other Air Emissions	-0.13	-0.22
Landfill Leachate		
Avoided External Costs From Energy Generation		
Electricity	1.01	5.18
CHP	1.45	7.44
Total external costs, no displaced burdens	-3.37	-5.05
Total external costs, displaced burdens from electricity	-2.36	0.13
Total external costs, displaced energy from CHP	-1.92	2.38
External Benefits from Nutrient Displacement	0.05	0.90
External Benefits from Pesticide Reduction	0.17	0.26
External Benefits from avoided nitrous oxide emissions	0.01	0.12
External Benefits from avoided process wastewater disposal	0.01	0.02
External Benefits from avoided peat extraction	0.23	0.37
Total external costs, no displaced burdens	-2.90	-3.39
Total external costs, displaced burdens from electricity	-1.89	1.79
Total external costs, displaced energy from CHP	-1.46	4.04
Memorandum Items		
Private savings from avoided fertiliser use	0.57	1.43
Private savings from avoided pesticide use	0.43	0.43

A.6.0 EXTERNALITIES OF TRANSPORT OF WASTE

INTRODUCTION

The external costs of waste transportation fall essentially into two categories:

a) work specific to waste transport; and

b) more general work on the externalities of transportation.

The range of external costs quantified includes:

1. Effects of air pollution (which, in respect of transport, has specific local, regional and global impacts) on health, agriculture and materials;

2. Contributions of transport to global warming;

3. Congestion;

4. Injuries and fatalities;

5. Road maintenance; and

6. Noise.

Different approaches to valuation are required to quantify these different externalities.

The changes which will be required to be implemented under the scenarios being modelled in this work include:

a) Changes in vehicles used (‘vehicle switching’);

b) Changes in distance travelled on collection rounds and between the round and the treatment facility; and

c) Changes in the proportion of the total distance travelled which occurs in urban and rural areas.

As argued in the main text, it is impossible to know with any accuracy what the nature of the changes required will be unless one understands the specific circumstances being investigated. Collection distances and distances travelled to treatment facilities vary considerably from country to country. The following sections are intended to give an indication of how important these factors are likely to be in the overall analysis.

AIR POLLUTION AND GLOBAL WARMING EFFECTS

The effects of air pollution depend broadly upon the effect of transport on pollutant concentrations and on the population exposed to the change in concentration. This means that:

a) the effects depend upon the quantity of pollutants emitted, which itself relates to the vehicle weight or load on the engine, the drive cycle or average speed of the vehicle and the vintage;

b) the effect is related to how the pollutants which are emitted are dispersed. This is usually estimated through use of models, and different models may be used to model the changes in air pollutant concentrations locally and regionally;

c) the effects increase at higher population densities. This means that journies in urban areas are likely (other things being equal) to cause greater damage to human health than those in more sparsely populated areas.

In what follows, we assess the externalities of waste transport through reference to two studies as well as a body of work drawn upon in reviews carried out therein:

1. The ExternE body of work (in particular, European Commission 1999c); and

2. Work undertaken for the Department of the Environment, Transport and the Regions in the UK on externalities from heavy goods vehicle (HGV) transport (NERA et al 2000).

Insights from the ExternE Program

In work under the ExternE programme, local air pollution modelling was carried out using the ROADPOL model whilst ECOSENSE was used to model regional scale air pollution effects. The following results were obtained from the country case studies carried out under ExternE:

France

In this study, only the car journeys are relevant to this study. This is due to the fact that cars could be used less if municipalities choose to separately collect yard waste as well as kitchen waste at the kerbside from householders. On the other hand, some may simply seek to collect kitchen waste requiring yard waste to be delivered to central collection points which, to the extent that this already occurs, implies no net change in external costs.

The French results suggest that depending upon the nature of the journey (predominantly rural or predominantly urban), the external costs per tonne of car travel are of the order €0.030 - €0.076 for a petrol car with a catalytic converter to €0.055 - €0.562 for a diesel car. The key difference between vehicles was the impact of particulates from the diesel fuelled car, which was much greater in urban areas.

Germany

The German study gave rather more by way of detail on the impacts across vehicles of relevance to this study. The Table below summarises the impacts of the transport modes. Note that these are for air and energy related emissions only. The first two columns are for emissions related to vehicle use only. The final column relates to fuel production, vehicle production, maintenance and disposal, and infrastructure.[30]

Table 76: Damage Costs for Transport Modes

Modes	Inter-city case (vehicle use)	Urban case (vehicle use)	Total from life-cycle emissions
	Per vkm	Per vkm	Per vkm
Car			
No catalyst	0.043	0.038	0.013
TWC old (pre 1987)	0.018	n.q.	0.013
TWC modern (before EURO2)	0.010	0.012	0.013
TWC new (EURO2)	0.008	0.010	0.013
Diesel old (1986-88)	0.030	0.065	0.010
Diesel modern (German standard 1996)	0.024	0.049	0.010
Lorries (all diesel)			
GVW <= 7.5t old (1980s)	0.165	0.355	0.016
GVW <= 7.5t modern (EURO1)	0.127	0.279	0.016
GVW 14-20t old (1980s)	0.313	0.795	0.031

[30] The ExternE report cites a study by Maibach (1995).

GVW 14-20t modern (EURO1)	0.239	0.615	0.031
GVW 20-28t old (1980s)	0.306	0.646	0.046
GVW >32t old (1980s)	0.479	1.041	0.069

Source: Adapted from Case Studies Germany in European Commission (1999c)

Note: The estimate of the uncertainty interval is given as a factor of 4-6 (it is not clear whether this is for the 68% or 95% confidence interval).

Greece

In the Greek study, emissions are reported for passenger cars using unleaded petrol and with a closed loop catalytic converter. The external costs are €0.017 / vkm in a residential street when the engine is cold falling to €0.009 when the engine is warm. On a motorway, the costs fall to €0.008 / vkm.

For a truck with 11.5 tonne load (not untypical of a 24t refuse collection vehicle when full), the damages vary enormously in urban and rural areas. In a major urban street, the figure is given as €3.515 / vkm falling to €0.431 / vkm on a motorway.

Netherlands

Results for the Netherlands case are given in the Table below.

Table 77: Damages (€/vkm) Associated With Transport in the Netherlands

	Damages in € / vkm			
	Across densely populated areas, case 1	**Across densely populated areas, case 2**	**Across moderately to sparsely populated areas**	**Across moderately populated areas**
Trucks (all diesel)				
3.5 – 16t (1990 built)	0.330	0.361	0.241	0.290
16t (1990 built)	0.534	0.660	0.345	0.420
Puller				
>16t (1990 built)	0.761	0.989	0.462	0.566
Average (1990 built)	0.576	0.721	0.367	0.448
Average (1998 built)	0.072	0.086	0.054	0.060

Source: Adapted from Case Studies: Netherlands in European Commission (1999c).

Note: The estimate of the uncertainty interval is given as a factor of 4-6 (it is not clear whether this is for the 68% or 95% confidence interval).

Insights from NERA and AEA Technology Report

Another study in the UK reported environmental costs as a combination of noise costs and environmental costs (NERA et al 2000). These figures are shown in Table 78. They are average costs so that the sensitivity to the area in which transport occurs is not captured in the Table. Noise is especially sensitive to population density in the modelling used in the study since results were calculated on the basis of a contingent valuation study reporting individuals' willingness to pay for reductions in noise pollution.

Elsewhere in the same report, some indication of the sensitivity of the environmental costs is given:

'Overall, in urban areas, the example values range from around 10 – 30 pence/km for Euro I vehicles travelling at medium speeds, from the smallest rigid to the largest articulated vehicle and so the difference between the lowest and highest damage is a factor of three[…] Damages from pre-Euro vehicles will be around one and a half times

as large (i.e. up to around 40 pence/km for the largest vehicle) and Euro II vehicles around half this value. However, at lower speeds, the values will increase further, and for very large artics, travelling at very low speeds in built up areas, the damage costs may be around 50 pence/km. Environmental costs are therefore potentially very significant relative to the internal costs of freight haulage operation.

However, the values decrease as the population density around the road decreases. For Euro I vehicles travelling at the same speeds in non-urban areas, the values drop to 4 – 11 pence/km (for smallest to largest Euro I vehicles). Again, there will be a higher and lower values for pre-Euro and Euro I vehicles, and for vehicles travelling at different speeds.' (NERA et al 2000)

Table 78: Environmental Costs (€ per vehicle-km)

Vehicle type	Axle configuration	Emissions standard		
		Euro II	Euro I	Pre-Euro
Rigids 3.5-7.5 tonnes	Mainly 2	0.0064	0.0075	0.0098
Rigids 7.5-12 tonnes	Mainly 2	0.0077	0.0093	0.0122
Rigids 12-13 tonnes	Mainly 2	0.0085	0.0102	0.0133
Rigids 13-14 tonnes	Mainly 2	0.0086	0.0106	0.0138
Rigids 14-15 tonnes	Mainly 2	0.0088	0.0107	0.0141
Rigids 15-17 tonnes	Mainly 2	0.0074	0.0104	0.0147
Rigids 17-18 tonnes	Only 2 axles	0.0075	0.0107	0.0149
Rigids 17-21 tonnes	Mainly 3	0.0080	0.0110	0.0152
Rigids 21-23 tonnes	Mainly 3	0.0083	0.0114	0.0158
Rigids 23-25 tonnes	Mainly 3	0.0086	0.0117	0.0162
Rigids 25-26 tonnes	Mainly 3	0.0088	0.0118	0.0163
Rigids 26-31 tonnes	4 axles	0.0096	0.0130	0.0181
Rigids 31-32 tonnes	4 axles	0.0098	0.0133	0.0182
Artics under 23 tonnes	Any axle trailers	0.0069	0.0090	0.0122
Artics under 23 tonnes	2-axle trailers	0.0075	0.0096	0.0130
Artics under 23 tonnes	3-axle trailers	0.0074	0.0094	0.0128
Artics 23-28 tonnes	Any axle trailers	0.0080	0.0102	0.0139
Artics 23-28 tonnes	2-axle trailers	0.0080	0.0102	0.0138
Artics 23-28 tonnes	3-axle trailers	0.0080	0.0102	0.0139
Artics 28-31 tonnes	Any axle trailers	0.0083	0.0107	0.0144
Artics 28-31 tonnes	2-axle trailers	0.0083	0.0107	0.0144
Artics 28-31 tonnes	3-axle trailers	0.0083	0.0107	0.0144
Artics 31-33 tonnes	Any axle trailers	0.0086	0.0109	0.0149
Artics 31-33 tonnes	2-axle trailers	0.0086	0.0109	0.0149
Artics 31-33 tonnes	3-axle trailers	0.0086	0.0109	0.0147
Artics 33-38 tonnes	2 + 2	0.0094	0.0117	0.0157
Artics 33-38 tonnes	2 + 3	0.0094	0.0117	0.0157
Artics 33-38 tonnes	3 + any	0.0094	0.0117	0.0157
Artics 33-38 tonnes	3 + 2	0.0093	0.0115	0.0154
Artics 33-38 tonnes	3 + 3	0.0094	0.0117	0.0157
Artics 38-40 tonnes	5 axles	0.0096	0.0120	0.0162
Artics 40-41 tonnes	6 axles	0.0099	0.0123	0.0165

Artics 41-44 tonnes	6 axles	0.0102	0.0128	0.0171

Source: NERA et al (2000).

Note how significant the change in vehicle emissions standards is in all cases for the damages from air pollution related to vehicle use. This suggests that ExternE results are already outdated to the extent that newer vehicles operate with lower emissions standards. New European emission directives exist for regulated pollutants from new vehicles manufactured or registered after 2000. For heavy duty vehicles, these are being introduced in two stages: Euro III (1 October 2001) and Euro IV (1 October 2006) standards. NOx emissions will be tightened up in a third stage, referred as Euro IV+, to be introduced on 1 October 2008.

Emissions will also be reduced by improvements in the quality of diesel fuel in accordance with the new European Fuel Quality Directives which take effect from 1 January 2000 and 1 January 2005. The main characteristics of diesel fuel that will change and affect emissions will be the sulphur content and density of the fuel. In the UK, ultra-low sulphur diesel, which virtually meets the Fuel 2005 standards, is already commercially available.

In addition, certain after-treatment emission abatement technologies can be retrofitted to HGVs, which are particularly effective in reducing particulate emissions. Opportunities for retrofitting most of these devices on existing HGVs are promoted by the early introduction of ultra-low sulphur diesel that the devices require for their effective operation[31]. The use of electric vehicles, or vehicles running on cleaner fuels, appears to be on the increase in waste management. Battery powered vehicles are beginning to be used to collect both compostables in Italy and recyclables in the UK (by no means everywhere in these countries).

The suggestion from these results is that the most modern waste management vehicles are likely to incur costs of approximately €0.006 - €0.010 per kilometre travelled, on average, with this being approximately doubled in urban zones. Given the way in which waste management vehicles travel in urban areas, the external costs might be expected to be towards, or beyond, the higher side of the quoted range. In urban areas, therefore, a figure of €0.03 might apply. This is still considerably lower than estimates from earlier studies based on vehicles with higher emission levels.

[31] It should be noted that the Euro IV and IV+ standards imply that the vehicle will be running on ultra-low sulphur diesel fuel which will be the fuel standard pertaining at the time the Euro IV vehicle emission standard comes into effect in 2005.

Congestion

When an extra vehicle joins a traffic flow at other than free flow conditions it imposes some additional delay on all other vehicles in the flow. In congested conditions, vehicles therefore impose external costs, in the form of slower journey time, on all other vehicles in the flow. This might be considered as a 'network externality', in which externalities are created by virtue of use made of an existing network.

In 'normal congestion', the relevant cost of congestion is the marginal cost, that is the extra cost imposed by an additional vehicle in the flow. This is equal to the sums of the delays imposed on all other vehicles in the flow. NERA estimated congestion costs in Great Britain, and subsequently extended the analysis to estimate the marginal congestion costs imposed by HGVs across the UK (based on Dodgson and Lane 1997 and Link et al 1999). The approach made use of two types of relationship:

- Speed-flow curves, which show a mathematical relationship between traffic flows on a road (usually in terms of vehicles per lane per hour), and the resulting traffic speeds on that road; and

- Operating cost formulae, which show a relationship – for a particular type of vehicle – between cost per km and speed. Since there is an exact inverse relationship between the speed and the time taken to travel one kilometre, time values can be incorporated into these formulae.

Combining these formulae gives a mathematical relationship between the volume of traffic flow on any section of road, and the cost of travel along that section of route. This means that it is possible to calculate the marginal cost of congestion for any traffic flow level.

Different vehicles make different contributions to congestion, eg an extra articulated goods vehicle slows down the average traffic speed more than does an extra car. This is usually allowed for by weighting vehicle numbers by 'passenger car units' (PCUS) where a car has a value of one, and other types of vehicle have values which reflect their relative contribution to congestion.

Congestion varies by region, by type of road and by time of the day and week. Consequently marginal congestion costs are likely to vary considerably. There are good reasons to believe that waste management vehicles are especially likely to impose congestion externalities upon other road users. The principal reason for this is that the collection of waste implies a stop-start drive cycle at slow speed in order to pick up waste materials.

We are not aware of any analysis that has sought to capture changes in congestion externalities arising from changes in waste management systems. We note the following here:

- Separate collection of compostable materials can enable a reduction in frequency of the collection of residual waste. Where this happens, the net change in congestion externalities is likely to be minimal;

- If no reduction in collection frequencies for residuals occurs, the net change in congestion externalities depends upon the factors mentioned above (region, type of road, and time at which the additional collection occurs). In addition, the size of vehicle and the collection approach (the logistics of the collection system) may play a part in determining the net outcome;

- The change in the journeys made between the collection of materials and delivery to a treatment facility will also have an impact, but these cannot be estimated with any degree of certainty and they are likely to vary significanty within and across countries.

Injuries and Fatalities

Road transport causes a number of accidents and fatalities each year. Statistically, these can be related to vehicle use. There is some discussion as to whether these costs are really ‘external’ since, to a significant degree (though by no means completely), they are borne by road users themselves (and might therefore be internalised in decisions as to whether or not to embark on journies).

One study combined data on major and minor accidents with estimates of values of statistical life to generate external cost figures for transport modes. The results are shown in Table 79.

Another study, using a different approach to generate the results and based in the UK, argued that the costs of accidents were to some degree internalised as costs to road users (also, through insurance premia etc.). It reported the results in Table 80 below (which have been converted to €). These are a factor of ten higher than the estimates for the UK quoted in the study above (for principal roads, they are a factor of 30 higher).

Table 79: External Costs from Transport (€ per 1000km)

	Mortality		Serious Injury	
	HGVs	**Passenger cars**	**HGVs**	**Passenger cars**

Belgium	6.59	1.26	0.17	0.08
Denmark	1.26	0.31	0.01	0.01
France	7.54	1.00	0.09	0.02
Germany	2.20	0.63	0.07	0.04
Greece	6.91	5.02	0.34	0.57
Irelands	1.26	0.63	0.01	0.01
Italy	2.51	0.63	0.23	0.14
Luxembourg	2.20	0.63	0.03	0.02
Netherlands	0.94	0.63	0.03	0.02
Portugal	7.22	0.63	0.64	0.79
Spain	4.71	1.57	0.25	0.20
UK	1.26	0.31	0.05	0.03

Source: PIRA et al (1998)

Table 80: Road Accident Cost per Vehicle Km (€)

Vehicles	Motorways	Other trunk	LA principal	LA other	All roads
Rigid					
Average cost	0.012	0.037	0.031	0.029	0.025
Articulated					
Average cost	0.012	0.035	0.025	0.022	0.020

Source: NERA et al (2000).

Road Maintenance

Vehicles running over transport infrastructure cause wear and tear on the roads. This depends upon axle configuration and the weight of the vehicle. Generally, a greater number of axles can compensate somewhat for the effects of increasing loads on the vehicle itself. Also influential are whether the vehicle is equipped with lift axles and whether it operates with dual tyres or 'wide single tyres' (the former being better from the perspective of reducing track wear). Estimates from a recent UK study are given in the Table 81 below.

SUMMARY

Total external costs from waste management vehicle transport should probably be calculated on the basis of a combination of the 'collection round' (where the vehicle stops and starts) and the 'transport to and from collection points'. Both of these should be split between rural and urban conditions, and on the basis of whether the journey is likely to be made at constant speed or in congested conditions. This makes the external costs of waste transport awkward to gauge (not least since different contributing factors depend upon different variables).

As an estimate of transport externalities, we have assembled Table 82 below for 7.5 tonne and 24 tonne vehicles. The air pollution externalities are representative of modern vehicles and we have taken average values from the studies above to be the 'low' values where waste management is concerned, and have used a factor of 2.5 (somewhat arbitrarily chosen, but intended to reflect the increase in externalities implied by waste transport in urban areas where the traffic moves slowly in densely populated areas) to illustrate 'high' values.

Table 81: Track Costs per Vehicle-Km

Vehicle type	Axle configuration	€ per vehicle-km
Rigids 3.5-7.5 tonnes	Mainly 2	0.0056
Rigids 7.5-12 tonnes	Mainly 2	0.0080
Rigids 12-13 tonnes	Mainly 2	0.0090
Rigids 13-14 tonnes	Mainly 2	0.0098
Rigids 14-15 tonnes	Mainly 2	0.0107
Rigids 15-17 tonnes	Mainly 2	0.0157
Rigids 17-18 tonnes	Only 2 axles	0.0173
Rigids 17-21 tonnes	Mainly 3	0.0136
Rigids 21-23 tonnes	Mainly 3	0.0160
Rigids 23-25 tonnes	Mainly 3	0.0227
Rigids 25-26 tonnes	Mainly 3	0.0269
Rigids 26-31 tonnes	4 axles	0.0283
Rigids 31-32 tonnes	4 axles	0.0339
Artics under 23 tonnes	Any axle trailers	0.0098
Artics under 23 tonnes	2-axle trailers	0.0098
Artics under 23 tonnes	3-axle trailers	0.0107
Artics 23-28 tonnes	Any axle trailers	0.0134
Artics 23-28 tonnes	2-axle trailers	0.0134
Artics 23-28 tonnes	3-axle trailers	0.0136
Artics 28-31 tonnes	Any axle trailers	0.0154
Artics 28-31 tonnes	2-axle trailers	0.0154
Artics 28-31 tonnes	3-axle trailers	0.0154
Artics 31-33 tonnes	Any axle trailers	0.0192
Artics 31-33 tonnes	2-axle trailers	0.0192
Artics 31-33 tonnes	3-axle trailers	0.0174
Artics 33-38 tonnes	2 + 2	0.0261
Artics 33-38 tonnes	2 + 3	0.0259
Artics 33-38 tonnes	3 + any	0.0256
Artics 33-38 tonnes	3 + 2	0.0256
Artics 33-38 tonnes	3 + 3	0.0198
Artics 38-40 tonnes	5 axles	0.0296
Artics 40-41 tonnes	6 axles	0.0230
Artics 41-44 tonnes	6 axles	0.0258

Note: these costs are derived on an average cost basis

NERA et al (2000).

Table 82: External Costs of Different Vehicles

Category of External Cost	7.5 tonne vehicle		24 tonne vehicle	
	Urban	Rural	Urban	Rural
Air pollution and noise	0.020	0.008	0.020	0.008
Injuries and Fatalities	0.035	0.020	0.035	0.020
Track Costs	0.008	0.008	0.013 / 0.024	0.013 / 0.024
Congestion	Nq	Nq	Nq	Nq
Total incl Injuries and Fatalities	**0.063**	**0.036**	**0.068-0.079**	**0.041 – 0.052**
Total excl. Injuries and Fatalities	**0.028**	**0.016**	**0.033 – 0.044**	**0.021 – 0.032**

These figures can be translated, roughly, into 'per kilometre per tonne' costs by assuming that approximately half the journey is made with a typical load, which for a 7.5 tonne truck might be 2.5 tonnes, and for a 24 tonne refuse vehicle, might be of the order 11 tonnes. Dividing the per vehicle kilometre figures by half these typical loads gives a per kilometre per tonne figure of the order €0.01 - €0.025 for a 7.5 tonne vehicle and €0.004 – €0.014 for a 24 tonne vehicle.

These external costs can be compared with the rates of fuel duty which internalise, however imperfectly, the external costs of transport in the countries under examination. Using assumptions regarding fuel consumption given below, and assuming an average load of half the typical payload for the vehicle, the fuel duty per kilometre per tonne can be calculated from rates of duty applicable to diesel fuels in the EU and other states (see Table 83). In all cases, the duties levied are of the same order of magnitude as the external costs quoted above. This suggests that there is a strong possibility that the external costs of transport are being internalised by various taxes and duties applied to diesel fuels. Although not all of these taxes are 'environmental', the suggestion is that the costs of transport externalities are already being paid for, though it has to be noted that not all the external costs of transport have been quantified here (notably the costs of congestion).

Our analysis tends to support the view that especially where the implementation of separate collection schemes is undertaken in such a way as to 'rationalise' the system of collection (through reducing frequencies of collection for residual waste), the need to account separately for any change in transport externalities is minimal. Furthermore, the analysis suggests that to the extent that these externalities were believed to be significantly non-zero, they are, in any case, substantially internalised in the financial costs of transport. To attribute additional financial costs as well as additional environmental costs risks 'double-counting' the effects of transport on the waste management collection system.

Table 83:Impact of Fuel Duty on Waste Transport (€ per km per tonne)

Vehicle	Fuel consumption (km /l)	Payload (t)				
7t van	**10**	**1.25**				
24t RCV	**7**	**5.5**				
	Austria	**Belgium**	**Denmark**	**Finland**	**France**	
7t van	0.0226	0.0232	0.0276	0.0260	0.0294	
24t RCV	0.0073	0.0075	0.0090	0.0085	0.0095	
	Germany	**Greece**	**Ireland**	**Italy**	**Luxembourg**	
7t van	0.0303	0.0200	0.0260	0.0323	0.0202	
24t RCV	0.0098	0.0065	0.0084	0.0105	0.0066	
	Netherlands	**Norway**	**Portugal**	**Spain**	**Sweden**	**Switzerland**
7t van	0.0277	0.0412	0.0197	0.0216	0.0291	0.0376
24t RCV	0.0090	0.0134	0.0064	0.0070	0.0095	0.0122
	UK	**Czech Republic**	**Estonia**	**Hungary**	**Poland**	**Slovenia**
7t van	0.0613	0.0177	0.0100	0.0237	0.0157	0.0264
24t RCV	0.0199	0.0057	0.0032	0.0077	0.0051	0.0086

A.7.0 BASELINE SCENARIO FOR MEMBER STATES AND ACCESSION STATES

INTRODUCTION

Different countries face different situations where the Landfill Directive Article 5 targets are concerned. Some had already diverted significant quantities of biodegradable municipal waste from landfill prior to 1995. We refer to these countries as 'high diverters.' There have been two prinicipal mechanisms for achieving this:

- through recycling of paper, wood and textiles, and the composting of municipal waste. The majority of states who have made significant progress in this direction appear to have achieved the latter through source separation of kitchen and garden waste, and collection of source segregated materials at containerparks / Civic Amenity sites. In turn, the vast majority of this material has been composted (rather than being subject to anaerobic digestion); and

- thermal treatment systems. The majority of states have some thermal treatment capacity, most obviously, incineration. There is, however, significant variation across Member States as to the degree to which incineration has been used to deal with residual waste. In some countries, with bans now in place on the landfilling of municipal waste (e.g. Netherlands and Denmark), residual waste will be dealt with principally by incineration. In others, such as Germany and Austria, the role of incineration is less prominent. In both these countries, landfilling of municipal waste is banned unless the waste has been sufficiently 'stabilised' (in the sense of having its fermentability reduced), leading to developments in mechanical biological treatment.

The relative emphasis on incineration and recycling / composting varies across this group of high diverters (of waste from landfill). For those who have proceeded furthest with composting and recycling, incremental increases in diversion through this means may be more difficult to achieve in future, though these countries (Germany, Austria, Netherlands, Flanders) tend to have in place measures likely to make recycling more 'possible' in future (through influencing the design of packaging via producer responsibility and consumer behaviour).

Within the group of high diverters as a whole (Austria, Belgium (Brussels and Flanders) Denmark, Germany, Luxembourg, Netherlands, Sweden), there are distinctions in the degree to which source separation is being emphasised as the principle mechanism for increasing, at the margin, the rate of diversion of biodegradable material from landfill in the future. In some countries, incineration seems likely to continue to increase. In others, the emphasis on source separation seems likely to remain much stronger, and it may well be that these countries make greater use of mechanical biological teatment (for example, Flanders is already proposing the construction of four such plants).

The ‘low diverters’, who still landfill the majority of waste produced, face a different problem. With the exception of Finland, which recycles a sizeable fraction of municipal waste, relatively little waste has been diverted from landfill through recycling and composting. As such, for the ‘low divertors’, of which France incinerates a sizeable fraction of waste, there is considerable scope for increasing recycling and composting. In theory, the potential for these countries to meet Landfill Directive targets through recycling and composting alone is likely to be greater than in those countries which had already made significant progress in recycling and composting prior to 1995. This is because, for the latter group of countries, the marginal increases in recycling and composting achievement are likely to be more difficult.

Growth rates, however, play a significant role in determining the feasibility of achieving the Landfill Directive targets through recycling and composting alone. Assuming that it is not possible to recycle / compost 100% of all biodegradable municipal waste, at high growth rates, meeting targets through recycling and composting alone becomes impossible (because targets concerning what can or cannot be landfilled are set relative to a 1995 baseline). On the other hand, constructing mass burn incinerators has the potential to undermine recycling / composting programmes almost before they have started in many municipalities.

In all countries, growth rates will play a key factor in determining costs for obvious reasons. It is interesting to note the range of purchasing power (per capita) across the States examined in this study. The range is enormous (even excluding Luxembourg, there is an almost four-fold difference in per capita wealth across the States – see Table 84). Given that there appears to be some correlation across Member States between wealth and waste creation (albeit, possibly less conclusive because municipal waste includes varying fractions of commercial waste as well – see Figure 6), the potential for growth is likely to be greatest in the poorest countries. Arguably, the way in which Landfill Directive targets are set penalises such countries. Ironically, however, Estonia and Czech Republic have some of the most challenging minimisation targets (250-300 kg per capita by 2010 and 340kg per capita by 2005, respectively).

FORWARD PROJECTIONS

Forward projections for the baseline ‘Landfill Directive Only’ scenario require projections of:

- Arisings (i.e. rates of growth in arisings);
- Composition and its evolution; and
- Likely future treatment options.

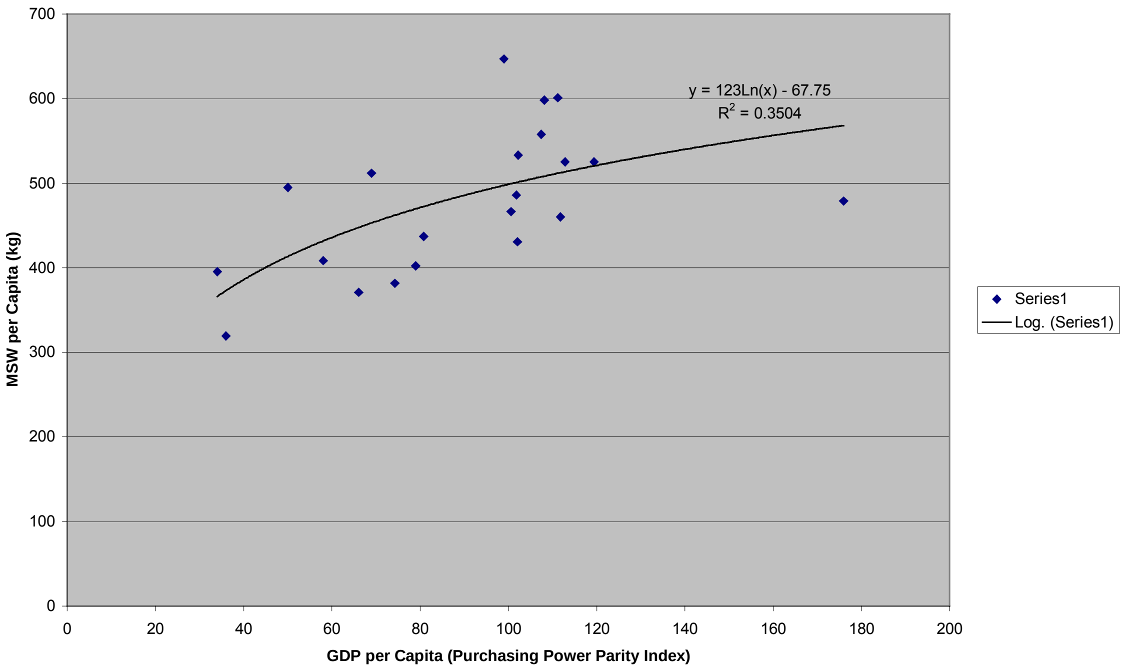
In the ideal world, the first two are well known, and the assignment of different waste fractions to specific treatment options follows a more or less well understood National Plan. In practice, growth rates are not easy to anticipate over the twenty year period we are looking at, and composition, though it is bound to change, will change in ways we can as yet only dimly perceive. We are crystal-ball gazing.

Table 84: Statistics On Municipal Waste Per Capita And Purchasing Power Of GDP

Country	Relative Purchasing Power Of Per Capita GDP	Municipal Waste Per Capita Per Year (kg)
Estonia	34	395
Poland	36	319
Hungary	50	495
Czech Republic	58	409
Greece	66	371
Slovenia	69	512
Portugal	74	382
Cyprus	79	402
Spain	81	437
France	99	647
Italy	101	466
Finland	102	486
Sweden	102	431
United Kingdom	102	533
Ireland	107	558
Germany	108	598
Austria	111	601
Belgium	112	460
Netherlands	113	525
Denmark	119	525
Luxembourg	176	479

Source: Population data and PPP index for Member States from Eurostat; for Accession States, population data is taken from the ongoing REC project on Waste Policies in Central and Eastern European Countries, and PPP index is from OECD and estimates (re-based to match EU figures).

Figure 6: Plot Of Per Capita MSW (kg) .v. GDP Per Capita (purchasing power index)



As regards treatment options, we have little option other than to assign treatments to the facilities as projected in National Plans, though even here there is some leeway, and for some countries, no clear plan is yet available (and this applies especially, though not exclusively, to the Accession Countries, where concerns with the plethora of Directives on waste are giving rise to major changes). Where no data is available, we have made what we believe to be plausible projections on the basis of the information we have.

Clearly, a 'health warning' is attached to all the scenarios which have been developed. We would urge commentators not to become too bogged down in detailed analysis of the projections, though we have tried to ensure that non of the projections are completely implausible. A brief discussion of the elements which make up the projections follows.

Growth Rates

Attempts at projecting changes in municipal waste arisings are fraught with difficulty. A brief review of experience can be found in a recent report for the European Environment Agency, as well as a somewhat heroic attempt to project forward on the basis of what the study itself admits is very shaky historic data (Christiansen and Fischer 1999).

Typically, projections are made on the basis of assumptions concerning the growth of GDP. Yet the European Topic Centre for Waste has been keen to explore, and promote, the concept of delinking waste arisings from economic growth (Christiansen and Munck-Kampmann 2000). The Topic Centre estimates that in European countries of the OECD, waste generation increased by an estimated 10 % between 1990 and 1995, while GDP increased by 6.5 %. However, it notes that at the Member State level, de-linking of municipal waste generation from household expenditure is being achieved in a few countries. The Netherlands, Iceland and Germany in particular, appear to show successful de-coupling of municipal waste generation from economic activity over time. Other states appear not to be making as much progress.

The report notes that household expenditure is not the ideal explanatory variable since municipal waste collection usually includes some commercial and industrial waste. Indeed, it has to be noted that one way in which many countries could, in theory, fulfil a substantial fraction of their Landfill Directive obligations would be to reduce collection of this waste, forcing it to be collected by private sector operators (or potentially, to simply re-define the term 'municipal waste'). To the extent that this is possible under prevailing legislation / obligations of local authorities, it becomes a cheap option to pursue. This is a clear problem with basing policy on what happens to what is essentially an administratively defined (if it is defined at all) category of waste.

The fact remains, therefore, that many factors will determine municipal waste arisings in the future. The discussion concerning delinking of arisings from economic growth is an interesting one. Forward projections made on the basis of the past risk entrenching views that such delinking cannot occur. Growth rates for municipal waste are, however, susceptible to influence by municipalities (for example, through limiting collection quantities, or through variable charging, consumer education, procurement policies), as well as national legislation, especially in respect of producer responsibility.

In this context, on the basis of our review, it seems possible to group countries into three broad categories:

- EU Member States which are setting serious objectives for the reduction, or reduced growth rates, for municipal waste;

- EU Member States which, having sought to understand the rate of growth in municipal waste, have projected forward on the basis of continuation of trend; and

- States which, despite concerns for what are, in some cases, rapid rates of growth in municipal waste arisings, appear to be torn between addressing this, and a desire to ensure that existing treatments (such as landfill) are of satisfactory standard whilst not hindering economic growth.

Even if growth rates may be known with some degree of certainty today, it may be that future rates are quite different. Our projections are made on the basis that States actually achieve what is broadly intended in their strategies though we have not included negative growth rates. This implies a strong interpretation of the ability of policy to achieve its stated objectives. On the other hand, it also gives due recognition to the efforts and intent of those countries which are seeking to address prevailing growth patterns, as opposed to those who seem to be taking a more fatalistic approach to waste arisings. It has to be recognised that policies on waste management across the EU are radically different. Up until now, this difference in approach has been highlighted through reference to the relative performance in recycling and composting (though home composting remains something of a 'black box' in this regard) in the different Member States, as well as the continued dependence upon landfill of States which have made less progress in this respect (though no state is without some dependence on landfill). In future, it seems likely that, as hinted at in the report by the ETCW referred to earlier, the difference in approach will manifest itself in widely varying growth rates (and it seems fair to say that these are already observable).

Table 85 presents some key information used in formulating the policy-off baseline. The Table also shows the extent to which we expect countries to be affected by the policy scenarios under discussion. What is interesting is that a number of countries are unlikely to be affected significantly by the implementation of a policy requiring source separation of Biowastes. For others, it seems less likely that their path to compliance with Landfill Directive requirements will co-incide with the requirements of a Biowaste Directive in terms of source separation.

The growth rates reported in the second column of Table 85 vary considerably. If one were to project these forward over a twenty year period, the evolution of per capita MSW arisings would be as shown in Figure 7. The Figure clearly illustrates the difficulties one faces in projecting forward over extended periods on the basis of current growth rates. Rather than converging over time as one might have expected (especially under increasingly divergent costs for waste treatment, and, potentially, economic convergence) the per capita figures diverge with Greece's figure being the greatest in the years beyond 2012. This reflects a general lack of understanding concerning the determining features of waste generation, itself a problem deepened by the lack of quality datasets.

On the one hand, one should recognise the efforts being made by those making serious attempts at minimisation. At the same time, projecting forward on the basis of existing growth rates is clearly unrealistic over an extended period.

Table 85: Basis For Country Projections

Country	Growth Projection	Laws / Aims	Destination of Diverted Biodegradable Wastes (from landfill)	Extent of Change Due to Policy On Source Separation
Austria	3.9% based on recent past	Must source separate / home compost Waste landfilled must be less than 5% volatile organic solids	Mostly source-separated composting / home compost and paper recycling – some AD, and some MBT post source separation. Possibly some incineration too	None?
Belgium	FL: 0.5% (plan is redn)	FL: Wide range of bans on landfilling and constraints on incineration B: W:	FL: Further source separation of paper / card, and home composting / collection of compostables B: W: Our impression is that W is following Fl approach	**FL: None** B: W: None ????
Denmark	0% (plan is stabn / redn)	Ban on landfilling of waste suited for incineration 30% recycling / composting of household waste by 2004, 40-50% in longer-term	Recycling of paper / board and probably split between compost/AD for garden waste, and some incineration. There may be requirements for source separation introduced (extent unclear)	Likely to reduce incineration and increase source separated compost, possibly increase in BMT too (depending upon incineration capacity)
Finland	In 2005, the amount of waste should be 15% less than the amount which would arise in the absence of reduction	Recovery of 70% of MSW by 2005, mostly through recycling, composting and AD. Recovery of 75% of biowaste by 2005 through composting and AD Recovery of 75% paper and card by 2005. From 2005, no MSW may be landfilled unless biodegradable fraction has been separated at source	Recycling of paper (some EfW) and composting / AD for putrescibles. National regulations on sorting of biowaste are envisaged. Sorting regulations will be introduced in municipal waste management regulations.	None? (if intentions of Plan are followed through)

Country	Growth Projection	Laws / Aims	Destination of Diverted Biodegradable Wastes (from landfill)	Extent of Change Due to Policy On Source Separation
	measures.			
France	2%	50% of municipal waste to be collected for recycling or composting (Circular 28/04/98). Ban on landfilling of 'ultimate' waste by 2002. Interpretation has evolved, and the leftovers from other kinds of treatment, such as advanced sorting and/or biological treatment, can be counted as 'ultimate' waste.	Moratorium on incineration proposed by Environment Minister – source separation of compostables and paper, with MBT to stabilise residual fraction, increasingly popular. If opinion / politics change, more incineration capacity could be developed. Some increase in mixed waste composting in Brittany possible.	Likely to increase source separation and home composting at the expense of incineration, mixed waste composting and landfill.
Germany	Slight reduction (1% over ten years)	Requirement to source separate. Only landfill material with less than 5% volatile solids, or material with 'equivalent' treatment (such as MBT).	Mainly source sep compost and AD, home compost, MBT and incineration (incremental paper recycling)	**Minimal / None**
Greece	Doubling in 10 years??	25% of BMW to be composted by 2005	MRF composting of mixed waste, SOME SOURCE SEPARATION (90:10) MRF recycling. Residual to landfill	Shift from MRF composting to source separation and possibly MBT for residual
Ireland	3.8%	None other than those required by the Landfill Directive. Considering a landfill levy.	Mix of source separated and mixed waste composting, anaerobic digestion, waste to energy incineration and thermolysis for biowaste and paper, and paper recycling. Public opinion likely to limit resort to EfW.	Shift from MRF composting / WTE to source separated composting / MBT
Italy	1-2%	National Law established minimum level of source separation of 35% by 2003. Pay as you throw, requirements for source separation, incentives for source sep and home compost (through landfill levies / charges for households) being established	Probably considerable source sep compost with paper recycling and MBT, some incineration / RDF manufacture	Some shift from 'integrated solutions' to source separation / MBT
Luxembourg		Organic components of MSW and comparable source have to be composted or treated, and a central aim is the separate	Source sep. / home composting / recycling	Minimal / none

Country	Growth Projection	Laws / Aims	Destination of Diverted Biodegradable Wastes (from landfill)	Extent of Change Due to Policy On Source Separation
		collection and treatment of organic waste.		
Netherlands	Targeted reduction appears around 1% p.a.	Compulsory for municipalities to collect organic household waste separately from January 1 1994. Ban on landfill of MSW other than in exceptional circumstances. Aim was 60% recycling by 2000	Source sep compost / AD, home composting, paper recycling, incineration of residual	None
Portugal	Reduce MSW by 2.5% by 2000 and by 5% by 2005 (from 1996 levels).	Reduction in MSW of 2.5% by 2000, and 5% by 2005; Recycling of MSW by 15% in 2000 and 25% in 2005; Composting of MSW of about 15% by 2000 and 25% by 2005;	MRF composting (integrated solutions with energy recovery), some source-separation and paper recycling	More source separation, less incineration, less mixed waste compost / integrated treatment
Spain	Increasing	Requirements for source separation (though not specifically for organic waste nationally)	Mixture of incineration, MRF composting, source sep composting and paper recycling	More source separation compost, less incineration / RDF
Sweden	3%	There is a requirement to separate combustible waste that will come into force in 2002, and a ban on landfilling of combustible waste will take effect at the same time. There will be a ban on landfilling organic waste in 2005.	Probably increase in source separated compost / AD (energy rec), recycling, and incineration (with heat recovery)	Our projection suggests 'not much' but likely to increase source separation and reduce incineration
UK	3%[a]	Targets for recovery / recycling and composting, tradable permits. Targets (in England only) are 40% of municipal waste recovered by 2005, 45% by 2010, 67% by 2015; Doubling the rate of recycling by 2003 (for English authorities), 25% of household waste recycled or composted by 2005; 30% recycled and composted by 2010; recycle or	Mix of MRF and source separated composting / AD / recycling and incineration / gasification	Increase in source sep composting and AD and MBT, less incineration / gasification

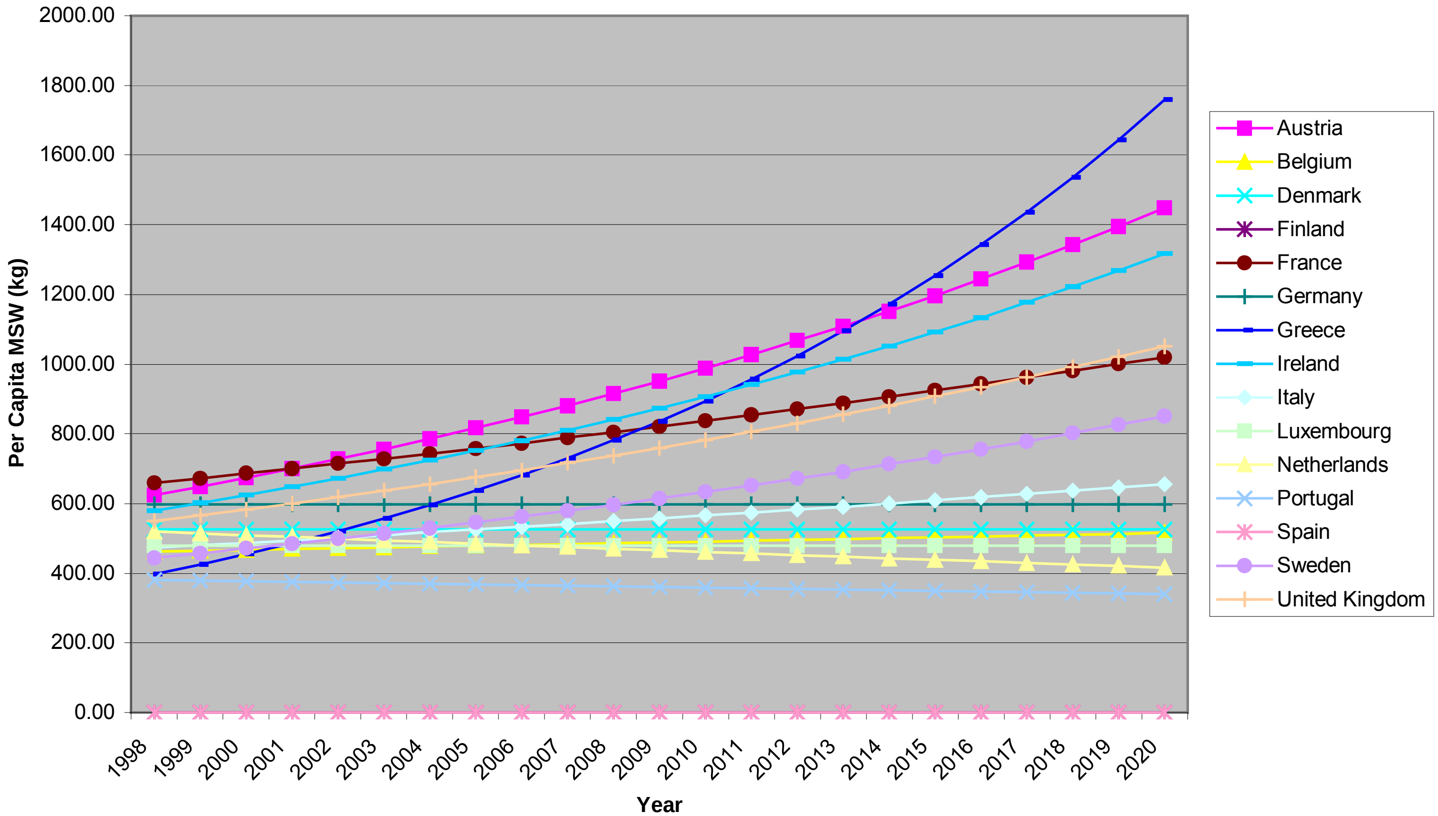
Country	Growth Projection	Laws / Aims	Destination of Diverted Biodegradable Wastes (from landfill)	Extent of Change Due to Policy On Source Separation
		compost 33% by 2015. Wales is likely to set targets for recycling and composting (from source separated materials) of 40% by 2010, possibly rising thereafter Scotland also seems likely to move further down the source separation route than England, though no targets have been set		
Cyprus	2.6%		NOT KNOWN – VERY LITTLE SOURCE SEPARATION AT PRESENT	NOT KNOWN
Czech Rep	Low – decrease recorded from 1998 to 1999	National Strategy aims for reduction in municipal waste per capita from 409kg to 340 kg by 2005. By 2005, compost 25% of compostable waste, recycle 65% of paper, incinerate 5% of municipal waste	ISPA strategy suggests emphasis on incineration and paper recycling, less on composting – 2 new 100ktpa plant suggested pre-2010 (EU funded)	Less incineration, large increase in source separation for biological treatment
Estonia	Approx 2% p.a. (avge.) since 1995 – seeking to reduce through e.g. packaging taxes	National Strategy targets for recycling – 30-40% by 2000 and 50% by 2010. Stabilisation of waste at low levels.	Emphasis on packaging waste recycling and minimisation. There are no MSW incinerators in Estonia at present	
Hungary	3-4% p.a. since 1995	Hungarian MoE states that targets for the separate collection of the biodegradable fraction of MSW will be included in the new Ministerial Decree, as this is the only possible way to produce good quality compost from MSW	Composting and recycling with some incineration??	May be limited but could reduce EfW and increase source separation / MBT?
Poland	2% p.a. between 1995 and 1999	No clear strategy as such. However. Incineration is not viewed favourably (there are 4 incinerators in Poland but only one with	More likely to be recycling and composting.	

Country	Growth Projection	Laws / Aims	Destination of Diverted Biodegradable Wastes (from landfill)	Extent of Change Due to Policy On Source Separation
		capacity >3t per hour		
Slovenia	0.5%	1995 Strategic Guidelines set targets for recycling and re-use of 35% of MSW by 2000, 55% by 2010. Source separation encouraged	Mix of composting, recycling and incineration	More composting based on source separation, less incineration

[a] It is generally accepted that this figure has been inflated by the effects of Landfill Tax drawing trade waste into the 'free-of-charge' municipal stream.

Source: The data in this Table was drawn from a range of different souces in the countries concerned.

Figure 7: Evolution Of Per Capita MSW In Member States



Estimates of growth in household waste from the European Environment Agency are shown in Table 37. This was based on a model estimated on the basis of data from Austria and the Netherlands. These do not appear to reflect the information we have gathered regarding municipal waste. Nor does it seem reasonable to ignore the effects of policies which, whilst they may be being harmonised in the EU, still vary significantly, especially in respect of efforts to reduce municipal waste growth (where there is currently no harmonisation)

Table 86: EEA Estimated Growth Rates for Household Waste

	1995-2000	2000-2005	2005-2010	1995-2010	Average Growth Rate (1995-2010)
BE	4%	5%	5%	15%	0.94%
DK	13%	10%	10%	36%	2.07%
FI	10%	6%	6%	23%	1.39%
FR	1%	-4%	4%	2%	0.13%
GR	3%	11%	12%	28%	1.66%
IT	3%	5%	5%	13%	0.82%
NL	9%	10%	10%	31%	1.82%
PT[1]	6%	9%	10%	28%	1.66%
ES[1]	8%	8%	8%	25%	1.50%
SE	9%	9%	9%	29%	1.71%
IE	20%	14%	9%	50%	2.74%
UK	11%	10%	11%	36%	2.07%
AT	4%	5%	6%	15%	0.94%
DE	8%	8%	8%	26%	1.55%
Total	7%	6%	8%	22%	1.33%

Source: Christiansen and Fischer 1999

1. Data on household waste not reported for PT and ES. Coefficient and projection estimates based on municipal waste data.

2. The estimated particular low growth of household waste in France is due to the relative share of the historical observed economic variables used to explain the development in the waste amounts compared to the overall GDP, and the continuation of this trend until 2010.

We propose to use rates of growth of 0%, 1% and 2% over the 20 year period with 1% as the central rate.

Composition

As regards composition, at first glance, it is extraordinarily difficult to justify anything other than a constant waste composition over time. Paradoxically, however, we know (or we think we do) that such an assumption must be fallacious. Not so very long ago, 'dustbins' consisted primarily of exactly that (dust), though with food waste besides once it was not composted at home (or in the community). The composition of waste over the 20 year period we are considering is bound to change.

As regards the biodegradable fraction, the key components are kitchen and garden waste, and paper. We discuss these briefly below.

Kitchen Waste

Kitchen waste might reasonably be assumed to be related to the amount of fresh fruit, vegetables and home consumed meals consumed in the household. However, the trend in many countries towards consumption of pre-prepared meals may be reducing the link between food consumption and generation of food wastes. A European Topic Centre report sought to link expenditure on food and beverages to waste arisings. Yet expenditure is probably a relatively poor predictor of waste quantities where food wastes are concerned (since expenditure is likely to switch towards higher value-added products rather than increased quantities). In addition, poorer families may consume more food in the home, and may prepare more of their own meals (pre-prepared meals are more expensive). Increased tele-working from home may lead to greater generation of food waste in households in the future (though equally, this may simply be displacing waste that would have been generated in the municipal stream, at commercial premises for example). There is reason to believe that the relation between socio-economic status and food waste generation is a complex one.

Recorded data will also be distorted by the role played by home / community composting. Where this activity increases in significance, other things being equal, the recorded data would show a reduction in the fraction of MSW accounted for by food waste. This may well be important for countries with large expanses of sparsely populated areas since local solutions such as these are likely to prove especially cost-effective means of dealing with food waste.

It may be that one other factor – expenditure on cut flowers – is of some significance in increasing waste generation of this type as wealth increases.

Yard Waste

Those without gardens will not generate yard waste. Ironically, in any areas where wealth is increasing, the effects of urbanisation have led to increased real estate costs and potentially, a reduction in the number of households who own gardens. One can speculate that the rate of growth in yard waste would be determined by:

- The rate of new-build housing in suburban areas; and

- The time and money spent by householders on gardening and related activity (though equally, households may also pave over gardens, for example, to provide patio space, or even garage space for cars).

Over a twenty year period, it is also possible that the effects of global warming could become observable. In much of Europe, one might assume that this could increase quantities of this type of waste either through changing climate over the year, or through the fertilisation effect of higher CO_2 concentrations.

It does seem difficult to believe that yard waste arisings could increase dramatically given these determinants, but without closer investigation, it is not possible to be certain about this.

As with kitchen waste, recorded data will be affected by the role played by home / community composting. Where this activity increases in significance, other things being equal, the recorded data would show a reduction in the fraction of MSW accounted for by garden waste. Home composting may, in turn, be affected by the nature of waste collection systems. Where the collection system includes the collection of garden wastes, this may reduce the inclination of householders to engage in home composting, potentially leading to the collection of wastes which might not have required collection since they might have been home composted.

Paper and Board

'Paper and board' is a relatively heterogeneous waste stream. A number of factors seem likely to influence arisings. Firstly, the change in work habits and the increase in ownership of home computers could increase the proportion of paper and board comprised of printing papers (though again, this could be just moving municipal waste that would have been collected from commercial entities to the household stream). The fate of newsprint in the wake of the advance of new information technologies appears not to be as stark as some have predicted and consumption continues to rise (and a number of studies show positive income elasticities of demand, though usually less than 1 for higher income countries (increasing at lower income levels).[32]

Estimates of paper and board packaging are difficult to anticipate. Internet purchases may see more packages delivered direct to households with a consequent increase in

[32] For a brief review of elasticities, see ECOTEC (1999).

packaging arising in the municipal waste stream. On the other hand, tighter legislation aimed at reducing packaging arising at source can be expected to have an influence.

Between 1995 and 2010, paper arisings are projected to undergo an increase of between 45% and 64% for the EU14 (Luxembourg excepted) according to a recent projection (Christiansen and Fischer 1999). The model gives enormous variation across Member States (from 239% for Greece to –1% for Sweden under one of the model specifications). This range must cast some doubt over the model’s predictive ability.

Summary

We know that waste composition will change over the next twenty years but we need a crystal ball to tell us how. Past experience is unlikely to provide a reliable indicator of future arisings, especially when the policy environment is so fluid. Furthermore, development in materials technology can be expected, whilst the increasing prevalence of user-pay, or pay-as-you-throw schemes seems likely to give householders incentives to reduce their waste arisings. Consequently, we have used the assumption that composition will remain constant, even though we know that it is likely to change.

TREATMENT OPTIONS

The treatment options on which projections are based are clearly driven by requirements to meet Landfill Directive targets. Where clear national policies / plans are available, we have assumed these will be reflected in actual behaviour. Where they are absent, we have sought to estimate the projection for the country concerned on the basis of our own knowledge. Note, however, that implementation of some plans in place is likely to be difficult owing to public opposition from the public to what is being proposed (for example, in respect of incineration capacity).

Country Classification for Modelling Purposes

Given the above considerations, and the information in Table 85, we propose the following country classification for modelling purposes. From the perspective of the model, this defines what treatment options are likely to be used to meet the Landfill Directive targets in the countries concerned:

Group 1 - Germany, Austria, Netherlands, Belgium (Flanders) (Amlinger estimates 65-80% coverage) and Luxembourg – here, there is unlikely to be any major change under either the Landfill Directive or the proposed Biowaste Directive from the current situation. The effect may be greatest in some larger cities where collection has been problematic in some countries. But if the proposed Directive makes exemptions for certain 'justifiable cases', it may well be that these countries experience little or no effect given the progress already made and the direction in which they are moving. We assume that the Landfill Directive will lead to further increases in source separation for recycling of paper and materials suitable for composting / digestion. Efforts in respect of the latter are likely to focus on kitchen wastes since yard wastes are already dealt with highly effectively in some countries (including Denmark in Group 2 – see below). Flanders appears to collect separately more than 90% of yard wastes. We also expect an increase in mechanical biological treatment as a means of pre-treating residual waste prior to landfilling or 'one-off' landscaping applications;

Group 2 - Finland, Belgium (Wallonia), Slovenia, Estonia, Sweden, Denmark, Spain, Italy, France, Poland – in these countries, there is an intention to develop separate collection, but this is not especially widely diffused at present. In these countries, we expect additional diversion of biodegradable wastes from landfill to be achieved through increased recycling of paper (though in Denmark, Sweden and Finland, this is already well-advanced), more source separation of compostables, and incineration. We anticipate that the switch from the Landfill Directive compliance trajectory to that required under a proposed Biowaste Directive would be principally one from incineration of unseparated biowastes to composting / anaerobic digestion of separately collected fractions, possibly with MBT used as source separation is developed (see below). This shift is unlikely to be extremely pronounced. In particular, Finland has set high targets for the recovery of both paper and kitchen and yard waste, with recovery understood to be principally recycling, composting and anaerobic digestion. Finland, therefore, could be a 'Group 1' country;

Group 3: Ireland, UK, Portugal, Belgium (Brussels), Czech Republic, Cyprus and Hungary. In these countries, relatively little separate collection of compostables is being undertaken at present. Landfill Directive targets likely to be met through a mix of incineration, paper recycling, and composting of yard waste. Some composting of source separated kitchen wastes may occur, but mixed waste composting and incineration are likely to be the most common routes for treatment of biowaste other than yard waste delivered to containerparks / civic amenity sites. The effect of a requirement for source separation would be a pronounced shift away from incineration / mixed waste composting towards composting / digestion of source separated waste and, possibly, MBT (used as pre-treatment whilst source separation develops – see below); and

Group 4: Greece. Very little separate collection is being undertaken at present and there appear to be no major pushes for this to occur in future. Landfill Directive compliance is likely to be pursued through mixed waste composting and paper recycling. There may be some incineration but public support for this is absent. The proposed policy change would require radical changes

For the purposes of the modelling, therefore, the following approach has been taken to designing the Landfill Directive scenario (the baseline in this project) and the effect of a policy mandating separate collection of biowastes:

- The Baseline 1995 data requirement to divert waste from landfill is derived from the estimated biodegradable fraction of municipal waste in 1995.
- Group 1: separate collection developed up to 80% of total kitchen and garden waste fraction, remainder dealt through incineration and MBT (in a 50:50 ratio up to required level of landfill diversion in line with Article 5 of the Landfill Directive):
- Group 2: separate collection developed to 60% of total kitchen and garden waste, remainder dealt through incineration and MBT (in a 75:25 ratio up to required level of landfill diversion in line with Article 5 of the Landfill Directive); and
- Group 3: separate collection developed to 35% of total kitchen and garden waste, remainder dealt with through incineration and MBT (in a 75:25 ratio up to required level of landfill diversion in line with Article 5 of the Landfill Directive).

For the purposes of this analysis, the whole of the UK is treated as a Group 3 country with Greece also in the same Group.

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