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FINAL REPORT

ASSESSMENT OF THE OPTIONS TO IMPROVE THE MANAGEMENT OF BIO-WASTE IN THE EUROPEAN UNION
ANNEX F: Environmental assumptions

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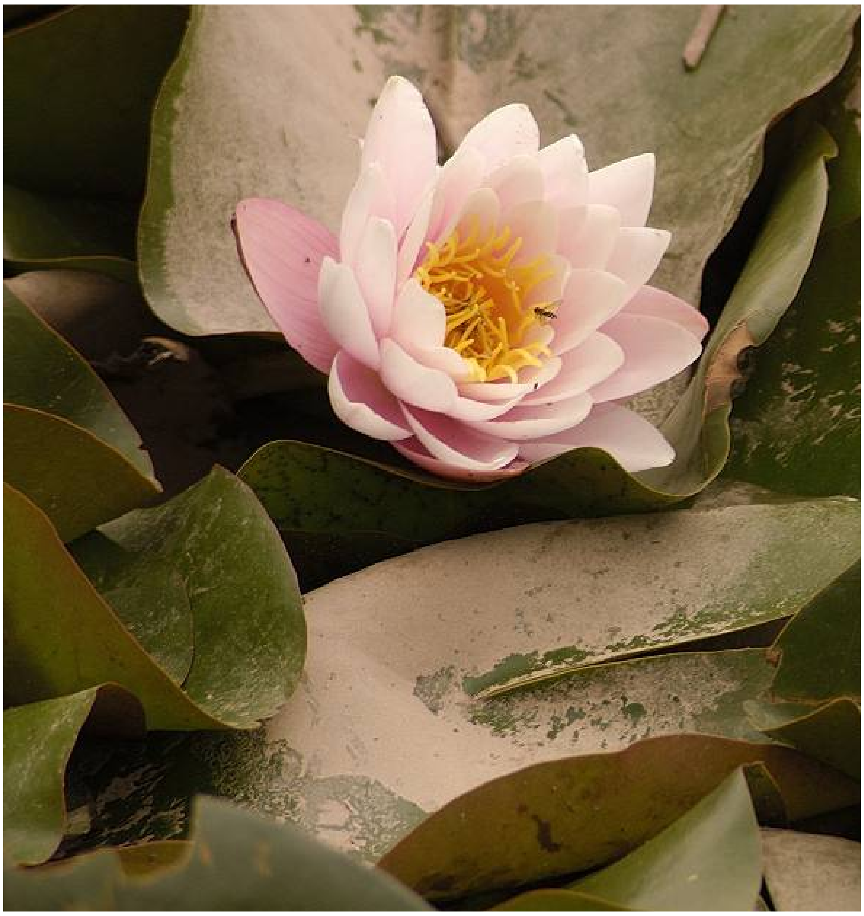

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ASSESSMENT OF THE OPTIONS TO IMPROVE THE MANAGEMENT OF BIO-WASTE IN THE EUROPEAN UNION

Infrastructuur, milieu, gebouwen

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A Generic Assumptions

A.1 Time

Time is an important factor when considering emissions modelling. Whilst incineration of biowaste results in an immediate release of CO_2, for example, composting biowaste with subsequent application to land results in at least partial sequestration of the organic carbon, with gradual release of CO_2 over an extended time period.[1]

If the overarching aim of any assessment is to determine the relative impacts of different technologies on climate change, and there is general consensus on the immediacy of the climate change issue, then the pace of release of greenhouse gases over time becomes an essential factor for consideration. In other words, the ability to sequester (or store) non-fossil carbon and effectively 'buy time' in terms of climate change is valuable. The importance of time-limited carbon sequestration was highlighted to the EU in a report by AEA Technology:[2]

> *However, for almost all treatment options, not all of the carbon released from organic materials during the treatment process is returned to the atmosphere; some remains in the 'residue' from the treatment process. This raises the issue of how this carbon should be accounted for, when comparing the treatment options in terms of climate change. If the carbon is sequestered in a form which is unavailable to the natural carbon cycle over a sufficiently long time period, then it could be argued that a 'sink' for carbon has been created and the treatment options should receive a carbon credit for this. The two main routes for carbon storage in waste management are in landfills (where the anaerobic conditions inhibit the decomposition of certain types of waste, particularly woody materials) and in compost applied to soil (where a proportion of the carbon becomes converted to very stable humic substances which can persist for hundreds of years). The permanency of such sinks is difficult to assess, and depends on the time scale used to define permanent. Available data suggests that 'woody' type materials in landfill may have only partially degraded over a one hundred year time scale, but degradation rates over a 500 year period are not known*.

LCA studies typically define a moment in time and aggregate all emissions occurring until that point in time. Such analyses have been criticised as not being a reliable indicator of the contribution of waste treatments to climate change because they ignore, to a certain degree, the dimension of time.[3]

For processes whose profile of emissions varies in time, this raises the following questions:

- Do emissions in all years count equally, or should a form of discounting be applied in such analyses? and;
- What is the justification for drawing the cut off in time in one year as opposed to another?

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[1] G. Finnveden, J. Johansson, P. Lind and A. Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, FMS: Stockholm

[2] AEA (2001) *Waste Management Options and Climate Change – Final report to the European Commission, DG Environment*

[3] Eunomia (2006) A Changing Climate for Energy from Waste? Final Report for Friends of the Earth, April 2006

In other words, 'doesn't time matter?' Given the discussion presented above regarding time-limited sequestration of non-fossil carbon, time evidently does matter, or at least should be considered in a comprehensive analysis.

Approach Taken in the Current Study

For the purposes of the present study we have applied the declining discount rate proposed by the UK's HM Treasury Green Book, as presented in Table A-1. The Green Book recommends using a discount rate of 3.5%. However, for projects with impacts exceeding thirty years, it recommends that a declining schedule of discount rates should be used rather a single, constant discount rate.

Table A-1: The declining long-term discount rate, as recommended in the Treasury Green Book

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

A.2 Biogenic CO_2 Emissions

A key issue in the assessment of GHG emissions from waste treatment technologies is whether or not non-fossil CO_2 (otherwise known as biogenic CO_2) should be included.

Under international GHG accounting methods developed by the Intergovernmental Panel on Climate Change (IPCC), non-fossil CO_2 is considered to be part of the natural carbon balance and therefore not a contributor to atmospheric concentrations of CO_2.[4] The rationale behind the IPCC's decision is that non-fossil carbon was originally removed from the atmosphere via photosynthesis, and under natural conditions, it would eventually cycle back to the atmosphere as CO_2 due to degradation processes. Climate change, however, is attributed to anthropogenic emissions, which impact this natural carbon cycle.

As regards waste, the Guidelines from IPCC state that the following should be reported: [5]

> *Total emissions from solid waste disposal on land, wastewater, waste incineration and any other waste management activity. Any CO_2 emissions from fossil-based products (incineration or decomposition) should be accounted for here but see note on double counting under Section 2 "Reporting the National Inventory." CO_2 from organic waste handling and decay should not be included.*

Specifically regarding waste incineration, the same guidelines state that reporting should include:

> *Incineration of waste, not including waste-to-energy facilities. Emissions from waste burnt for energy are reported under the Energy Module, 1 A. Emissions from burning of agricultural wastes should be reported under Section 4. All non-CO_2 greenhouse gases from incineration should be reported here as well as CO_2 from non-biological waste.*

Given the above, then it is worth reporting what is set out regarding energy. The following are to be reported:

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[4] Intergovernmental Panel on Climate Change. *Greenhouse Gas Inventory Reference Manual: Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories*, Vol. 3, Pg. 6.28, (Paris France 1997).

[5] Understanding the Common Reporting Framework, in IPCC (u.d.) Revised 1996 IPCC Reporting Guidelines for National Greenhouse Gas Inventories, Reporting Instructions (Volume 1), Hadley Centre, Bracknell

Total emissions of all greenhouse gases from all fuel combustion activities as described further below. CO_2 emissions from combustion of biomass fuels are not included in totals for the energy sector. They may not be net emissions if the biomass is sustainably produced. If biomass is harvested at an unsustainable rate (that is, faster than annual regrowth), net CO_2 emissions will appear as a loss of biomass stocks in the Land-Use Change and Forestry module. Other greenhouse gases from biomass fuel combustion are considered net emissions and are reported under Energy. (Sum of I A 1 to I A 5). Incineration of waste for waste-to-energy facilities should be reported here and not under Section 6C. Emissions based upon fuel for use on ships or aircraft engaged in international transport (1 A 3 a i and 1 A 3 d i) should, as far as possible, not be included in national totals but reported separately.

Methane (CH_4) is also derived primarily from non-fossil carbon during degradation processes. However, CH_4 emissions from landfills are counted within GHG inventories. The rationale provided by the IPCC can be described as follows:[6]

CH_4 emissions from landfills are counted - even though the source of carbon is primarily biogenic, CH_4 would not be emitted were it not for the human activity of landfilling the waste, which creates anaerobic conditions conducive to CH_4 formation.

Currently, convention appears to be shaped by IPCC's approach to dealing with non-fossil carbon in the reporting of Greenhouse Gas Inventories by different countries.

The crucial point here is that for the purposes of IPCC reporting, non-fossil CO_2 from incineration is effectively not reported – an approach also recommended by the French waste management industry.[7] Although it could be argued that this convention of ignoring non-fossil CO_2 is appropriate within the inventory context, it has perhaps erroneously been applied to comparative assessments between waste management processes.[8]

Whatever the merits or otherwise of not reporting biogenic CO_2 for the purpose of national inventories, in comparative assessments between processes, it cannot be valid to ignore biogenic CO_2 if the different processes deal with biogenic CO_2 in different ways. Given that different processes often deal with non-fossil CO_2 in different ways, and that the atmosphere does not distinguish between molecules of greenhouse gas depending on their origin, the omission of non-fossil CO_2 from analyses appears dubious. The need to include biogenic CO_2 is well recognized by some of those involved in life-cycle assessments, such as Finnveden *et al*.:[9]

The practise to disregard biotic CO2-emissions can lead to erroneous results (Dobson 1998). Let us consider an example to illustrate this. Let us compare incineration and landfilling of a hypothetical product consisting of only cellulose. When incinerated, nearly 100 % of the carbon is emitted as CO2. However, in the inventory, this emission is often disregarded as noted above. If the product is landfilled, approximately 70 % of the material is expected to be degraded and emitted during a short time period, mainly as CO2 and CH4 (Finnveden et al. 1995) (The short time period is here defined as the surveyable time period). Again the

[6] USEPA (2004) *Greenhouse Gas Emission Factors for Municipal Waste Combustion and Other Practices*

[7] L'Entreprises pour L'Environnement, *Protocol for the quantification of greenhouse gas emissions from waste management activities*, September 2006, Nanterre, France

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

[9] G. Finnveden, J. Johansson, P. Lind and A. Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, FMS: Stockholm

> *emitted CO_2 is normally disregarded, although the CH_4-emissions are noted. During the surveyable time period, 30 % of the carbon is expected to be trapped in the landfill. There is thus a difference between the landfilling and the incineration alternatives in this respect, in the incineration case all carbon is emitted, whereas in the landfilling case some of the carbon is trapped. This difference is however not noted, since the CO_2-emissions are disregarded and this is in principle a mistake. Additionally, the biological carbon emitted as CH_4 in the landfilling case is noted and will discredit this option. It could be argued that a part of the global warming potential, corresponding to the potential of the same amount of biological carbon in CO_2, should be subtracted from the landfilling inventory.*

Recent articles published in both the International Journal of Life Cycle Assessment and Science also recommend the same approach as that taken by Finnveden et al.[10]

The IPCC Guideline regarding emissions related to energy requires further analysis in the context of refuse-derived fuels (RDF). If the biomass portion of RDF is included under the definition of 'biomass fuels', then whether or not CO_2 emissions should be included (for inventory purposes) would appear to depend on the sustainability of the production of that biomass. Considering the heterogeneous mix of biological material contributing to the biomass portion of waste, the task of determining what is or is not sustainably produced would be extremely difficult. Should a comparison of the GHG intensity of waste management processes relative to traditional fossil fuel generation be undertaken, this might be a worthy approach.

In the IPCC Guidelines, in theory, this would not be of significance if one was confident that the reporting of inventories under the Agriculture, Forestry and Other Land Use (AFOLU) Section took adequate account of all the effects of waste-related activities on changes in soil carbon, carbon in the existing forest stock, etc. Using, as a convention, the assumption that the non-fossil CO_2 is unimportant risks, however, ignoring the matter of the potential significance of changing the rate of flux of CO_2 from non-fossil sources into the atmosphere. Clearly, burning biomass leads to the immediate release of CO_2. However, composting biomass leads to the production of compost which, on application to soil, increases the carbon stock, and releases the carbon over an extended period of time.[11]

Approach Taken in the Current Study

The current study includes all biogenic CO_2 emissions from waste management processes. Our approach to the biogenic CO_2 emissions resulting from wood combustion (where wood is used as a renewable energy source) is discussed in Section A.4.4.2.

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[10] See, for example: Rabl A, Benoist A, Dron D, Peuportier B, Spadaro J V and Zoughaib A (2007) How to Account for CO_2 Emissions from Biomass in an LCA, *Int J LCA*, 12(5) p 281; Searchinger T D, Hamburg S P, Melillo J, Chameides W, Havlik P, Kammen D M, Likens G E, Lubowski R N, Obersteiner M, Oppenheimer M, Robertson G P, Schlesinger W H and Tilman G D (2009) Fixing a Critical Climate Accounting Error, *Science*, 326, pp527-528

[11] See E. Favoino and D. Hogg (2008) The Potential Role of Compost in Reducing Greenhouse Gases, *Waste Management Research,* 2008; pp. 26; 61

A.3 Damage Costs for Pollutants

A.3.1 Monetised Climate Change and Air Quality Impacts

We have considered the impacts upon air quality that are expected to result from the treatment processes, including both direct and indirect impacts (the latter relating to avoided impacts associated with energy generation and the recycling of materials).

Our approach is to apply external damage costs to emissions of a range of air pollutants, allowing for the quantification of impacts in monetary terms.

The analysis that follows is focussed upon emissions to air. Whilst waste treatment processes may also in some cases affect soil and water quality, data regarding the precise nature of these impacts is less robust. Similarly, we have not included damage costs associated with disamenity of waste treatment facilities, as estimates of disamenity costs vary considerably between the different sources, and are not yet available for the MBT treatment processes.

The set of data that we have used has been extracted from the Clean Air for Europe (CAFÉ) programme and the Benefits Table (BeTa) database from the European Commission DG Research MethodEx project.[12] These datasets cover a wide range of pollutants (see Table A-2), and present country specific damage costs for non-GHGs.

Unit damage costs have been determined from the CAFÉ data, which is from 2000. In order to model in 2009 prices we have converted these costs into real 2009 figures using the Harmonised Index of Consumer Prices (HICP).[13] A number of member states have been omitted from this report and for these countries we have assigned the lowest unit cost for each pollutant. Table A-2 shows the costs that have been attributed to each member state for each pollutant. For the remaining pollutants that have been modelled the unit damage cost is constant across all member states. The pollutants and the modelled cost are shown in Table A-3.

[12] M. Holland and P. Watkiss (2002) Benefits Table Database: Estimates of the Marginal External Costs of Air Pollution in Europe, Database Prepared for European Commission DG Environment, database available at www.methodex.org/BeTa-Methodex%20v2.xls; AEAT Environment (2005) Damages per tonne Emission of PM2.5, NH3, SO2, NOx and VOCs from Each EU25 Member State (excluding Cyprus) and Surrounding Seas, Report to DG Environment of the European Commission, March 2005. The figures used reflect the Mean Values of Life Years approach to valuation, including health sensitivity. For CO2e the BeTa MethodEx central figure (€19/tonne in 2000 prices, €23/tonne in 2009 prices), was used.

[13] Eurostat (2009) HICP - all items - annual average inflation rate – (tsieb060). Available at http://epp.eurostat.ec.europa.eu/tgm/table.do?tab=table&init=1&plugin=1&language=en&pcode=tsieb060. An index was created using the values from 2000 to 2008, for the row entitled European Union. This gives an overall rate of inflation from 2000 to 2009 of 22.57%. Therefore damage costs were uplifted by 22.57% to give 2009 prices.

Table A-2: Variable CAFÉ Unit Damage Costs (€ 000's/tonne) in 2009 Prices

Member State	NH3	VOCs	PM2.5	SO2	NOx	Cd	Cr	Ni
AT	29 €	5 €	88 €	20 €	20 €	404 €	53 €	6 €
BE(Flanders)	74 €	6 €	147 €	26 €	11 €	674 €	88 €	11 €
BE(Wallonia)	74 €	6 €	147 €	26 €	11 €	674 €	88 €	11 €
BE(Brussels)	74 €	6 €	147 €	26 €	11 €	674 €	88 €	11 €
BG	4 €	0 €	10 €	3 €	2 €	47 €	6 €	1 €
CY	4 €	0 €	10 €	3 €	2 €	47 €	6 €	1 €
CZ	48 €	3 €	76 €	20 €	17 €	343 €	45 €	6 €
DK	20 €	2 €	40 €	12 €	10 €	184 €	25 €	3 €
EE	7 €	0 €	10 €	4 €	2 €	47 €	6 €	1 €
FI	5 €	0 €	13 €	4 €	2 €	60 €	8 €	1 €
FR	28 €	4 €	107 €	20 €	17 €	490 €	64 €	8 €
DE	43 €	5 €	116 €	27 €	22 €	539 €	70 €	8 €
EL	8 €	1 €	21 €	3 €	2 €	94 €	12 €	1 €
HU	27 €	2 €	61 €	12 €	12 €	282 €	37 €	4 €
IE	6 €	2 €	36 €	12 €	9 €	159 €	21 €	3 €
IT	27 €	3 €	81 €	15 €	13 €	368 €	49 €	6 €
LV	7 €	1 €	21 €	5 €	3 €	98 €	12 €	2 €
LT	4 €	1 €	21 €	6 €	5 €	93 €	12 €	1 €
LU	61 €	7 €	99 €	23 €	20 €	454 €	60 €	7 €
MT	20 €	1 €	22 €	5 €	2 €	47 €	6 €	1 €
NL	54 €	5 €	147 €	32 €	15 €	699 €	91 €	11 €
PL	25 €	2 €	70 €	13 €	9 €	319 €	42 €	5 €
PT	9 €	1 €	54 €	8 €	3 €	245 €	32 €	4 €
RO	4 €	0 €	10 €	3 €	2 €	47 €	6 €	1 €
SK	34 €	2 €	49 €	12 €	12 €	221 €	29 €	4 €
SI	31 €	4 €	54 €	15 €	16 €	245 €	32 €	4 €
ES	11 €	1 €	45 €	10 €	6 €	208 €	27 €	3 €
SE	15 €	1 €	28 €	7 €	5 €	135 €	17 €	2 €
UK	42 €	3 €	89 €	16 €	8 €	417 €	54 €	6 €

Table A-3: Unit Damage Costs from BeTa MethodEx Database, Constant in 2009 Prices

Cost per Tonne	Dioxin	Pb	Hg	CO2e
€/tonne	€ 45,350,900,000	€ 735,420	€ 7,354,200	€ 23

Note: Thousand separators as commas.

A.3.2 Omissions from the Analysis

The following is a list - almost certainly not extensive - of externalities not covered or not explicitly accounted for by the study (in all cases, the omissions relate to 'direct' and 'avoided' impacts).

Disamenity (including Odour and Nuisance)

The argument that there is insufficient data available to incorporate disamenity in a cost-benefit study comparing landfill with incineration is losing credibility. None of the impacts assessed can be said to have been estimated with a high level of certainty. The standard of proof for disamenity should not, arguably, have to be any higher. However, in comparing a wider range of treatments, we have chosen to omit disamenity from the analysis. Our view is that if disamenity was to be included, incinerators located in dense urban areas would fare worst, and well-managed biological treatment facilities (including quality odour treatment) in rural areas would fare best.

Air Emissions other than the following:

- CO_2
- CH_4
- N_2O
- NH_3
- VOCs
- $PM_{2.5}$
- SOx
- NOx
- CO
- Cd
- Cr
- Hg
- Ni
- Pb
- Dioxin
- As

Bioaerosols

It seems likely that the main risks relate to composting and anaerobic digestion where the digestion process includes a post-digestion aerobic step. However, some bioaerosols are likely to be present at all waste facilities. Key uncertainties which remain to be considered relate to the source factor as it relates to biowaste treatments (and, importantly, how the source factors might be reduced) and the exposure response relationship between the micro-organisms which may be released and the population exposed.

Emissions to Land

No emissions to land have been included other than in respect of incinerator fly ash residues. This almost certainly means that the treatment of landfills is too favourable. Where the application of source separated organic material to land is concerned, no environmental disbenefit is assumed to occur on the basis that application rates would not lead to elevated (above prevailing levels) concentrations of potentially toxic elements. A limitation of life cycle assessments is that the blanket application of toxicity weightings to applications of trace concentrations of metals to soil – even where the impact on soil quality is suggested by reputable studies to be very limited – tends to suggest major impacts in terms of toxicity where none may apply.

Emissions to Water

The emissions to water are likely to be a greater issue for landfills and for anaerobic digestion plants (depending upon the details of the facility's design and operation). It is sometimes assumed, in cost-benefit analyses, that the environmental costs of emissions to water are reduced, and effectively internalised, through payments for / investments in waste water treatment at waste management facilities. Our analysis includes some estimates of emissions related to treatment of leachate from landfills. No other emissions are accounted for. At compost facilities, it is assumed that the majority of water is re-circulated in the plant. At anaerobic digestion facilities, whether or not, and to what extent, process waters are treated at waste water treatment plants varies. Finally, pollution of water courses can be an issue for incinerators, notably where wet scrubbers are used, but in the facility modelled in this study, water pollution is not assumed to be a major issue.

Externalities Associated with Construction

We have not considered external costs associated with construction of facilities. It is generally stated that these account for a small proportion of the overall impacts. However, it is difficult to be quite so sanguine about this when a cost-benefit perspective, incorporating non-zero discount rates, is employed. All construction-related externalities occur early in time (by definition). Consequently, the construction related externalities will weigh proportionately greater in an analysis using discounting than in one where no discounting is used (for example, in most life cycle assessments).

Household Time

The effect on household time has not been considered in this study. The reader is referred to a recent study for a discussion of this issue.[14]

Water use at Facilities

This is another impact which is not captured in conventional life-cycle assessment. We have looked at the effect of compost applications on reducing the requirement for irrigation water. We have not looked at the use of water at the plants themselves. Demand is likely to be greatest at incinerators, though equally, low solids AD facilities will require considerable water. The degree to which process waters can be recirculated in the process is likely to vary (for all plant types) with detailed design.

Land Use

Some studies have debated whether or not to assess the opportunity cost of land in the assessment of externalities.[15] Generally, however, the view tends to be adopted that land values are reflected in the cost of facilities.[16] This has not been included in this analysis.

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[14] See D. Hogg (2006) *Impact of Unit-based Waste Collection Charges*, Report for the OECD Environment Directorate, Working Group on Waste Prevention and recycling, May 2006.

[15] For example, a Dutch study attributed significant externalities to landfill related to these costs (E. Dijkgraaf and H. R. J. Vollebergh (2004), Burn or Bury? A Social Cost Comparison of Final Waste Disposal Methods, *Ecological Economics*, 50, 233-247).

[16] This includes most, if not all, studies other than that cited in the previous footnote, including a recent Dutch study which discussed Dijkgraaf and Vollebergh's approach (H. Bartelings, P. van Beukering, O. Kuik, V. Linderhof, F. Oosterhuis, L. Brander and A. Wagtendonk (2005) *Effectiveness of Landfill Taxation*, R-05/05, Report Commissioned by Ministerie von VROM, November 24, 2005

Transport

Transport affects the analysis only insofar as the options lead to significant changes in the use of transport fuel. Some models of the waste sector have, in the past, assumed that specific distances relate to the use of specific facilities. We would suggest that this assumption is not justifiable.

Evidently, the various options being considered in this study could lead to changes in transport logistics and changes in waste miles. However, the relationship is a complex one. If, when recycling collections are introduced, the logistics of refuse collection are also affected, then depending upon which vehicles are used for which purpose, introducing additional recycling services can make very little difference to the actual fuel use associated with the new services.

Furthermore, as regards inland vehicular transport, transport externalities might be considered to be more or less fully internalised in fuel and other transport-related duties. The degree to which this assumption holds good relates to the significance accorded to (and the approach to valuing) congestion externalities, if indeed these should be considered as external costs. It can be argued that a proportion of the costs associated with congestion are not 'external' insofar as transport decisions are made on the basis of some knowledge as to when congestion is likely to occur. Indeed, in the waste management case, service providers will be sensitive to congestion-related issues in terms of the timing of their collection rounds. However, if marginal congestion costs are estimated on the basis of marginal additions to traffic, these can be quite high, and the assumption that existing duties internalise all externalities almost certainly breaks down.

Notwithstanding these points, to the extent that the assumption might not be valid, then to the extent that one is seeking to understand changes in the transport externalities across different systems, it can reasonably be argued that these changes are unlikely to have a major influence on the analysis.

A.4 Energy Use by Member States

A.4.1 Marginal Sources of Electricity and Heat Generation

Within the field of life cycle analysis, a distinction is made between consequential and attributional approaches to evaluating the environmental impacts of a system or product.[17] The attributional approach aims at describing the environmentally relevant flows to and from a life cycle and its subsystems. Such an approach would typically use an average energy mix to determine the impacts. In contrast, the consequential approach aims at describing how the environmentally relevant flows from the technological system will change in response to possible changes in the life cycle. The latter approach uses the marginal source of generation to evaluate environmental impacts resulting from the change.

Given that the scenarios appraised in this study reflect changes to current waste management, the use of the marginal energy source appears to be the right approach to take within the current analysis. Determining which is the marginal source across each of member states is not, however, straightforward.

In a report to the UK Department for Business, Enterprise and Regulatory Reform (BERR) investigating compliance costs for meeting the 2020 Renewable Energy Target,

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[17] T Ekvall and B Weidmema (2004) System Boundaries and Input Data in Consequential Life Cycle Inventory Analysis, *International Journal of Life Cycle Analysis*, 9(3), pp161-171

Poyry assumed the marginal source of electricity generation across all the member states to be based on gas, their justification for this assumption being consistency with previous BERR analyses.[18]

In their international review of the greenhouse gas emissions from energy systems, Dones et al proposed that the marginal electricity source should reflect the latest available technology or mix of technologies (e.g. the cheapest available for base load), or those technologies which will most probably cover an increase in demand.[19] This echoes the view of Defra in the UK who suggested the marginal technology should be that which was the new build capacity, indicated to be Combined Cycle Gas Turbine (CCGT) for the UK.

Many countries are increasingly reliant on gas for both heat and electricity generation. However, energy use statistics produced by Eurostat confirm that in 2006, some member states – notably Cyprus and Malta - did not use any gas for either heating or electricity generation.[20] In Sweden, France, Poland and Slovenia, the proportion of electricity generated from gas was less than 5% of the total generational capacity in 2006, whilst the proportion of heat generation from gas was less than 10% of the total for Finland, Greece and Sweden. Other countries such as Germany and Romania use significantly more coal than gas as their fuel source for electricity generation.

Analysis published by RWE in 2007 presented details of proposed new electricity plant for a number of member states.[21] Their report covered planned electricity generation capacity over 300 MW that were expected to be in place by 2012. Whilst the authors noted that gas projects were dominant, a significant quantity of additional coal or lignite generation capacity was suggested for Germany, Bulgaria, the Czech Republic and Poland.

Choosing the marginal source of heat might be considered a simpler proposition than for electricity. There is, however, limited analysis in the literature relating to avoiding emissions from heat production. For heat, more so than electricity, Profu emphasizes the importance of local conditions in determining what should be modelled as the marginal source.[22] This suggests that the choice of marginal heat source should be made with specific reference to local drivers and constraints, rather than with reference to demand.

Approach Taken in the Current Study

In the absence of detailed statistics confirming either the new build electrical generation capacity for each of the member states or the local conditions influencing heat supply, we have chosen to base our analysis of the impacts associated with electricity and heat generation upon the average generation mix for each country for both types of energy. We recognise that this will increase the impacts associated with energy generation for some countries. Sensitivity analysis is therefore carried out which confirms the impact upon the results of using gas as the marginal source of generation for the electricity and heat.

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[18] Poyry (2008) Compliance Costs for Meeting the 20% Renewable Energy Target in 2020: A Report to the Department for Business, Enterprise and Regulatory Reform, March 2008

[19] Dones R., Heck T., Hirschberg S., (2004) “Greenhouse Gas Emissions from Energy Systems, Comparison and Overview.” In: Encyclopedia of Energy (Ed. Cleveland C.), Vol. 3, pp. 77-95. Academic Press/Elsevier, San Diego, USA

[20] Commission of the European Communities (2008) Second Strategic Energy Review: Europe’s Current and Future Energy Position, Part B - Statistical Annex, Report to the European Parliament

[21] RWE (2007) Factbook: Generation Capacity in Europe, June 2007

[22] Profu, *Evaluating waste incineration as treatment and energy recovery method from an environmental point of view*, CEWEP, May 2004

The electricity and heat mix statistics presented in the following sections are combined with the emissions factors in Section A.4.4 to obtain the overall emissions factors for heat and electricity generation in each member state. A summary of the energy related impacts for each country is presented in Section A.4.4.4.

A.4.2 Electricity Mix

Table A-4 presents the electricity mix for each of the 27 member states in 2006, using data provided by Eurostat.

Table A-4: Electricity Generation Mix – EU Member States

	Oil	Gas	Nuclear	Re-newables	Solid fuel	Other
Austria	3%	19%	0%	62%	11%	5%
Belgium	2%	30%	54%	4%	8%	2%
Bulgaria	1%	5%	42%	9%	42%	1%
Cyprus	100%	0%	0%	0%	0%	0%
Czech Rep.	0%	5%	31%	4%	59%	1%
Denmark	4%	21%	0%	22%	53%	0%
Estonia	0%	8%	0%	1%	90%	0%
Finland	1%	16%	28%	27%	27%	1%
France	1%	4%	79%	11%	4%	1%
Germany	2%	12%	26%	12%	42%	6%
Greece	16%	17%	0%	13%	53%	1%
Hungary	2%	37%	38%	4%	20%	0%
Ireland	10%	51%	0%	9%	29%	1%
Italy	15%	52%	0%	17%	14%	2%
Latvia	0%	43%	0%	57%	0%	0%
Lithuania	3%	20%	69%	3%	0%	5%
Luxembourg	0%	75%	0%	25%	0%	0%
Malta	100%	0%	0%	0%	0%	0%
Netherlands	2%	60%	4%	10%	24%	0%
Poland	2%	3%	0%	3%	91%	1%
Portugal	11%	25%	0%	32%	31%	1%
Romania	3%	19%	9%	29%	40%	0%
Slovak Rep.	2%	7%	58%	15%	17%	1%
Slovenia	0%	3%	37%	25%	36%	0%
Spain	8%	30%	20%	17%	22%	3%
Sweden	1%	1%	47%	50%	1%	0%
UK	1%	36%	19%	5%	38%	1%

Source: Commission of the European Communities (2008) Second Strategic Energy Review: Europe's Current and Future Energy Position, Part B - Statistical Annex, Report to the European Parliament

A.4.3 Heat Mix

Table A-5 presents the heat mix for each of the 27 member states in 2006, developed from data provided by Eurostat.

Table A-5: Heat Generation Mix – EU Member States

	Oil	Gas	Renewables	Solid fuel
Austria	35%	29%	21%	15%
Belgium	47%	35%	5%	13%
Bulgaria	26%	28%	9%	38%
Cyprus	89%	0%	6%	5%
Czech Rep.	16%	29%	7%	48%
Denmark	31%	32%	28%	9%
Estonia	10%	23%	18%	48%
Finland	35%	8%	39%	18%
France	42%	36%	12%	9%
Germany	36%	37%	9%	18%
Greece	66%	6%	9%	19%
Hungary	20%	60%	8%	12%
Ireland	51%	27%	4%	18%
Italy	36%	44%	10%	10%
Latvia	12%	39%	46%	3%
Lithuania	28%	48%	18%	6%
Luxembourg	33%	53%	5%	10%
Malta	100%	0%	0%	0%
Netherlands	37%	51%	5%	7%
Poland	19%	21%	8%	52%
Portugal	53%	13%	29%	5%
Romania	24%	47%	12%	18%
Slovak Rep.	17%	47%	4%	32%
Slovenia	38%	28%	16%	18%
Spain	51%	28%	10%	11%
Sweden	33%	4%	49%	14%
UK	28%	56%	3%	13%

Source: Commission of the European Communities (2008) Second Strategic Energy Review: Europe's Current and Future Energy Position, Part B - Statistical Annex, Report to the European Parliament

A.4.4 Environmental Impacts of Energy Use

There can be considerable variation in the environmental impacts of energy generation. Much of the variation is associated with the choice of fuel – for example, the impacts (in terms of emissions to air per kWh of electricity generated) associated with the use of renewable sources of electricity and nuclear are typically far less than those associated with oil and coal. However there is also variation between emissions resulting from use of the same fuel to generate the same type of energy in different circumstances. The latter includes:

- For fossil fuels, there may be natural variability associated with the fuel itself. This may have an impact on pre-combustion emissions, which will affect emissions associated with both heat and electricity generation. Important examples of this include:

- Mining coal in some countries causes substantial release of CH_4, depending on the location of the mine;[23]
- The sulphur content of natural gas, high levels of which are typically flared (increasing the SOx emissions associated with pre-combustion);[24]

- Emissions associated with fuel transportation can vary considerably between member states, particularly for gas which may be transported across Europe for thousands of kilometres to countries that do not have their own gas supply. Leakage from these long distance pipelines can be significant, causing pre-combustion emissions associated with the use of gas to increase;[24]
- Variations associated with the use of combustion and generation technology, notably:
  - Improved efficiency of electrical generation leading to a reduction in CO_2 emissions per kWh;
  - The same improved efficiency may, however, cause an increase in other emissions. As gas turbine efficiencies increase, the resultant higher firing temperatures lead to greater NOx formation. Thus measures taken to reduce NOx formation will be less effective at the larger and more efficient CCGT plant.[25]

Some indication as to the scale of one of these sources of variation was provided by Poyry in their recent report to the UK's BERR investigating compliance costs for meeting the 2020 Renewable Energy Target. They suggested country specific emissions factors for gas electricity generation of between 397-410 grams CO_2 equivalent per kWh, taking their estimates from the National Greenhouse Gas Inventory Reports.[26] It is not clear whether these estimates included the pre-combustion emissions for each member state.

Given such a diverse sources of variation, it is difficult to develop country specific emissions factors with any degree of certainty. It is anticipated however that the energy mix will have the most significant impact on the energy impacts for each member state. Our approach has therefore been to develop "typical" emissions factors and use these factors across all member states.

A.4.4.1 Electricity

Table A-6 confirms the emissions factors used to estimate the impacts of electricity generation for the different generation sources considered within the current analysis.

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[23] D. Weisser (2007) A Guide to Life-cycle Greenhouse Gas Emissions from Electric Supply Technologies, Energy, 32, pp1543-1559

[24] ecoinvent (2004) ecoinvent Data v1.1, Final Reports ecoinvent 2000, No. 1-15, Swiss Centre for Life Cycle Inventories, Dubendorf, 2004

[25] European Commission (2006) *Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for Large Combustion Plants,* July 2006

[26] Poyry (2008) Compliance Costs for Meeting the 20% Renewable Energy Target in 2020: A Report to the Department for Business, Enterprise and Regulatory Reform, March 2008

Table A-6: Emissions Factors for Electricity Generation

	Emissions factors, kg / kWh electricity				
	CO2e	**NMVOC**	**$PM_{2.5}$**	**SOx**	**NOx**
Gas	0.4	2.70E-04	1.40E-05	2.20E-04	7.20E-04
Oil	0.8	4.05E-04	2.44E-04	6.86E-03	2.35E-03
Coal	0.9	1.21E-04	2.81E-04	4.52E-03	2.82E-03
Nuclear	0.0	2.58E-03	1.58E-05	3.67E-05	3.98E-05
Renewables	0.0	5.65E-06	1.30E-05	1.58E-05	3.26E-05

Notes

Includes pre-combustion emissions for the fossil fuels and nuclear (e.g. emissions associated with fuel extraction)

Electricity from renewables is assumed to be 80% hydro-electricity and 20% wind

Sources: Eurostat (2009) Panorama of Energy: Energy Statistics to Support EU Policies and Solutions; ecoinvent (2004) ecoinvent Data v1.1, Final Reports ecoinvent 2000, No. 1-15, Swiss Centre for Life Cycle Inventories, Dubendorf, 2004; D. Weisser (2007) A Guide to Life-cycle Greenhouse Gas Emissions from Electric Supply Technologies, Energy, 32, pp1543-1559

A.4.4.2 Heat

Table A-7 confirms the emissions factors used to estimate the impacts of heat generation for the different generation sources considered within the current analysis.

Table A-7: Emissions Factors for Heat Generation

	Emissions factors, kg / MJ heat				
	CO2e	**NMVOC**	**$PM_{2.5}$**	**SOx**	**NOx**
Gas	0.2	3.63E-05	1.61E-06	3.16E-05	4.51E-05
Oil	0.3	1.00E-05	1.00E-05	3.29E-04	1.50E-04
Coal	0.4	6.10E-05	3.48E-05	6.59E-04	2.14E-04
Renewables	0.1	3.47E-05	5.40E-05	1.03E-05	2.15E-04

Notes:

Includes pre-combustion emissions for the fossil fuels (e.g. emissions associated with fuel extraction).

The factor for renewables is based on the emissions factor for wood fuel and assumes two thirds of the fuel was sustainably produced (with regard to the CO_2 emissions).

Sources: Eurostat (2009) Panorama of Energy: Energy Statistics to Support EU Policies and Solutions; ecoinvent (2004) ecoinvent Data v1.1, Final Reports ecoinvent 2000, No. 1-15, Swiss Centre for Life Cycle Inventories, Dubendorf, 2004

The emissions factor for renewable heat generation assumes wood as the fuel source. Biogenic CO_2 emissions resulting from wood combustion are typically ignored in analyses that take their greenhouse gas accounting approach from the IPCC inventory, where emissions associated with the use of biomass fuels are not recorded in the totals for the energy sector. Section A.2 confirms that the use of such fuels will not necessarily result in net CO_2 emissions if the biomass is sustainably produced. It is however, impossible to tell how much of the wood used for heat generation across EU is obtained from sustainably managed sources, and the proportion of this is likely to vary between the different member states. Our approach is therefore to assume two thirds of the wood fuel combusted comes from sustainably managed forests across all member states with regard to the CO_2 equivalent emissions.

A.4.4.3 Diesel

We have used a figure of 3.26 kg CO_2 equivalent per litre of diesel (including 0.46 kg CO_2 equivalent pre-combustion emissions).

Data regarding the air quality impacts associated with the use of diesel within waste management facilities is taken from the BUWAL life-cycle inventory database produced by the Federal Office for the Environment in Switzerland.[27] The major air quality impacts are emissions of NOx (estimated to be 105 g per litre of diesel combusted), SOx (5 g per litre) and PM_{10} (2 g per litre).

A.4.4.4 Summary of Impacts for the Different Member States

Table A-8 shows the external costs for the utilisation of energy for each Member State, taking into account both the generation mix for electricity and heat (previously shown in Table A-4 and Table A-5 respectively) and the external costs associated with the pollutants for each country as discussed in Section A.3.

[27] Available from http://www.bafu.admin.ch

Table A-8: External Costs for Energy Utilisation (Average Energy Mix)

	External costs for electricity, € / MW		External costs for heat, € / GJ	
	Climate change	Air Quality	Climate change	Air Quality
Austria	€ 4.99	€ 33.67	€ 1.78	€ 9.03
Belgium	€ 4.70	€ 31.15	€ 1.72	€ 9.04
Bulgaria	€ 9.49	€ 77.93	€ 1.97	€ 1.58
Cyprus	€ 18.63	€ 224.34	€ 1.83	€ 1.47
Czech Republic	€ 12.89	€ 28.04	€ 2.06	€ 10.30
Denmark	€ 13.71	€ 31.06	€ 1.48	€ 3.80
Estonia	€ 19.79	€ 149.02	€ 1.96	€ 2.20
Finland	€ 7.45	€ 37.15	€ 1.57	€ 1.67
France	€ 1.41	€ 5.10	€ 1.68	€ 6.46
Germany	€ 10.35	€ 25.57	€ 1.80	€ 9.72
Greece	€ 15.79	€ 124.27	€ 1.88	€ 1.60
Hungary	€ 8.07	€ 62.81	€ 1.75	€ 3.23
Ireland	€ 12.81	€ 31.04	€ 1.85	€ 4.83
Italy	€ 11.18	€ 53.56	€ 1.80	€ 4.59
Latvia	€ 4.31	€ 15.11	€ 1.34	€ 1.12
Lithuania	€ 2.56	€ 13.77	€ 1.63	€ 1.56
Luxembourg	€ 8.38	€ 20.77	€ 1.95	€ 6.52
Malta	€ 18.63	€ 71.74	€ 1.86	€ 2.08
Netherlands	€ 10.71	€ 75.16	€ 1.61	€ 7.63
Poland	€ 19.75	€ 50.40	€ 2.09	€ 7.25
Portugal	€ 11.05	€ 106.35	€ 1.57	€ 2.46
Romania	€ 10.62	€ 55.05	€ 1.62	€ 1.05
Slovak Republic	€ 4.64	€ 16.95	€ 1.95	€ 4.99
Slovenia	€ 7.85	€ 17.63	€ 1.77	€ 6.20
Spain	€ 9.04	€ 46.16	€ 1.76	€ 3.55
Sweden	€ 0.44	€ 5.08	€ 1.45	€ 2.71
UK	€ 11.25	€ 46.52	€ 1.60	€ 4.05

A.4.5 Proportion of Heat Utilised by Facilities

Heat generation from waste management facilities is generated continuously and would not always be capable of being utilised. We have incorporated a heat load factor into the model, and this was set at 60% for the modelling runs in this report.

B Composting

Our model considers two types of composting process:

1. Open Air Windrow composting facilities;
2. In vessel (Enclosed) composting facilities.

These two types of process are discussed together within the sections that follow.

B.1 Direct Emissions to Air from the Process

B.1.1 Factors Affecting Plant Performance

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

1. The nature of the input wastes, in particular:
    a. the nature of the organic carbon in the components of the waste, which determines the biodegradability of the material (and hence, the extent of biodegradation over a give period of time); and
    b. the nature of any organic compounds in the input wastes which may be released as the mass of material heats up;
2. The nature of the process, and the retention time in that process, as well as the maturation period;
3. The nature and effectiveness of the turning / airflow systems, and the frequency of turning;
4. The regime of management of moisture in the biomass, especially in turned windrow systems;
5. The C:N ratio of the biowaste; and
6. The nature and effectiveness of any measures to control air pollution. Implicitly, this means that gaseous emissions from windrow facilities will be higher for some gases than they will be at enclosed facilities making use of biofilters.

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

B.1.1.1 Nature of Process, Retention Time and Maturation Period

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

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

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

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

Relatively few studies have approached this issue with the care it might seem to deserve. Only Vogt et al appear to have sought to distinguish, for the purpose of emissions, between fresh and mature compost. However, the effects of emissions in the process, and emissions after application to land, appear to largely cancel each other out. More important is the nature of the material which the compost substitutes when it is applied.

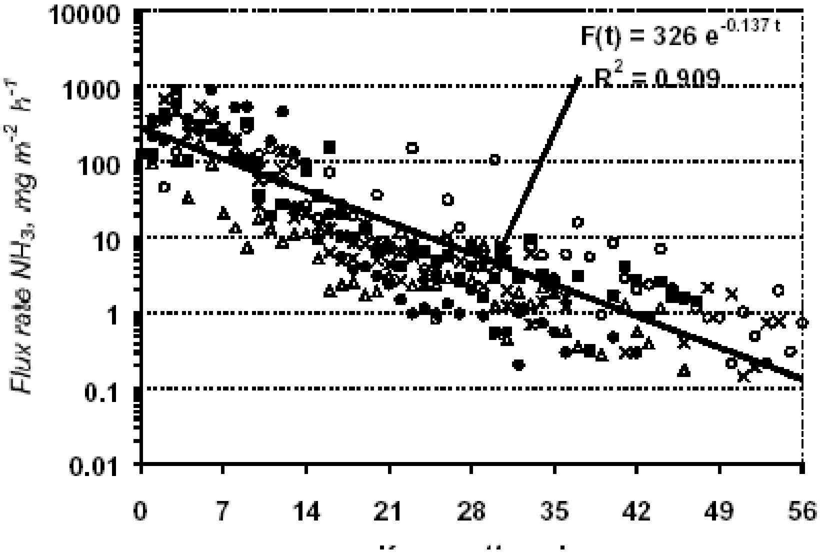
As regards laboratory measurements, one study (Hellebrand and Kalk, 2000) did seek to understand the evolution of the key compost-related emissions over time, but the feedstock was manure (as opposed to municipal biowastes).[28] An important observation of the study (to the extent that it may be transferable) was:

> *Over the course of the composting period, the emissions of ammonia and methane are reduced almost completely during the first three weeks. Nitrous oxide emissions exhibit great variation over the entire composting period. As the quantity of NH_3 and CH_4 diminishes, N_2O emissions show an increasing tendency. Maximum values are measured after two to six weeks. Afterwards, N_2O formation slowly decreases.*

Highest ammonia emissions were shortly after the manure was added to the process, and detection limits were reached after eight weeks (see Figure 1). Discussions with process managers appear to confirm that with municipal feedstocks as well, ammonia is predominantly released early in the process.

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

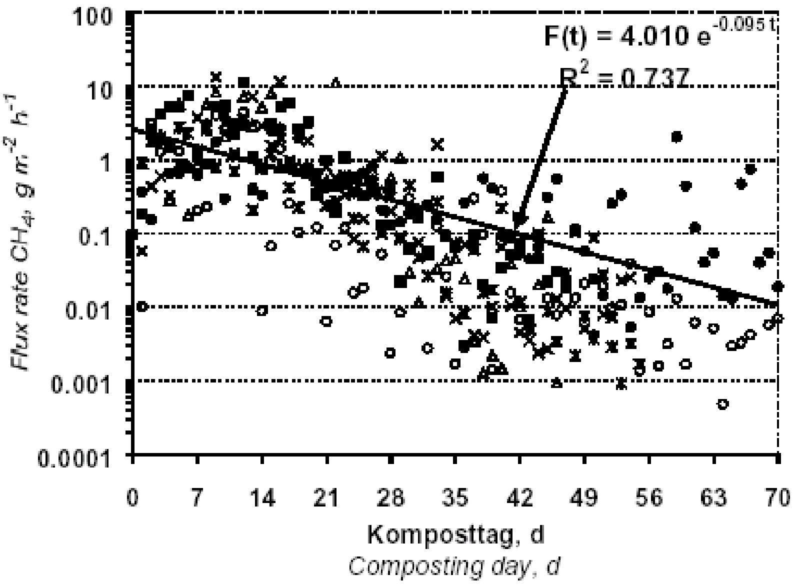
Figure 1: NH_3 Emissions with Exponential Trend Function for a 56 Day Composting Period



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

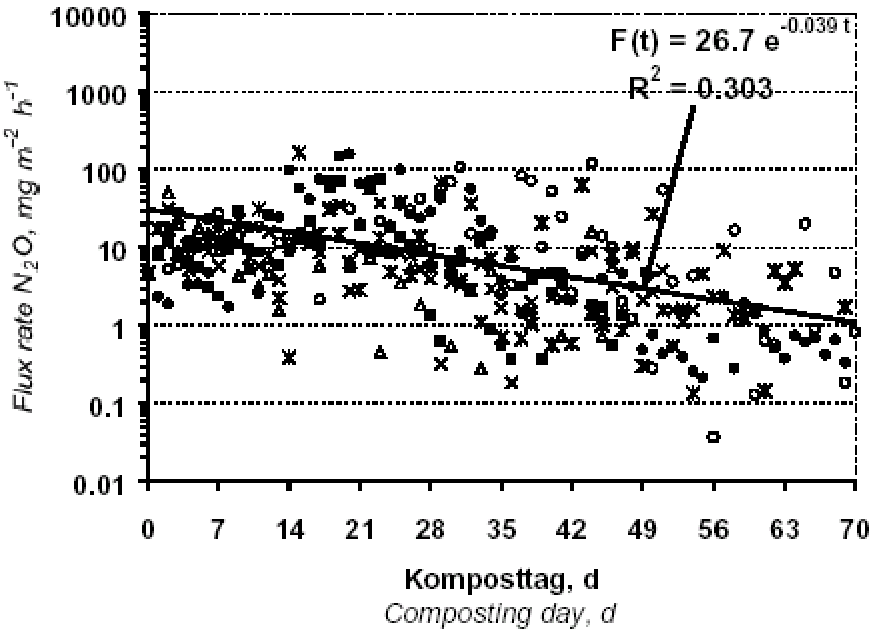
A similar function was used to describe methane emissions (see Figure 2). However, nitrous oxide emissions were more variable, exhibiting a less stable pattern of decay over time (see Figure 3).

Figure 2: CH_4 Emissions with Exponential Trend Function for a 70 Day Composting Period



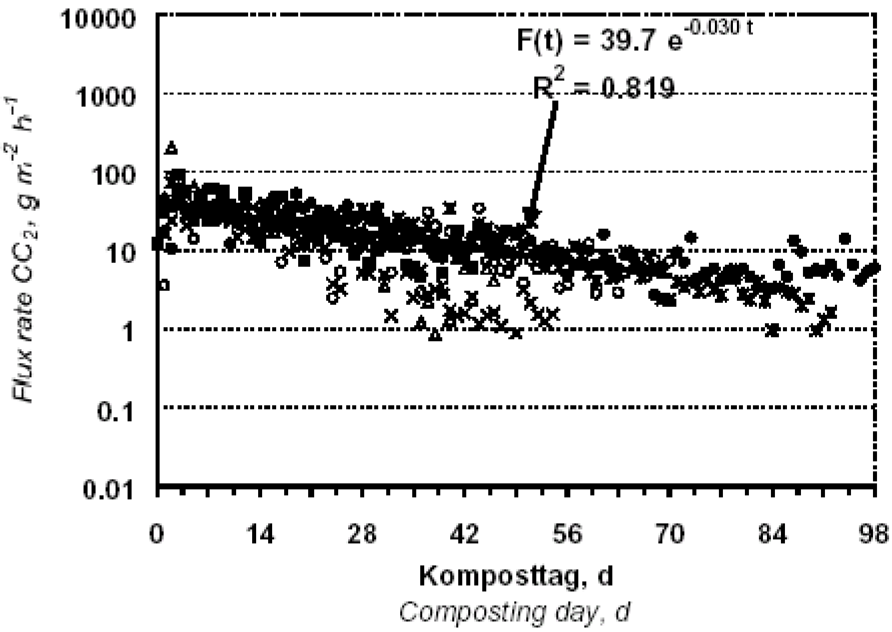
Source: Hellebrand, H. J. and W. D. Kalk (2000) Emissions Caused by Manure Composting, Agrartechnische Forschung 6 (2000) Heft 2, S. E 26-E 31

Figure 3: N_2O Emissions with Exponential Trend Function for a 70 Day Composting Period



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

Figure 4: CO_2 Emissions with Exponential Trend Function for a 70 Day Composting Period



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

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

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

B.1.1.2 Use of Biofilter / Scrubber

In most life cycle studies, the presence or otherwise of a biofilter is typically seen as important in respect of the impact categories of acidification and eutrophication. For the same reason, to the extent that these impacts are deemed problematic, the assumed efficiency of biofilter removal is of some significance.[29] Indeed, in many processes, the likelihood of significant problems arising in respect of either acidification or eutrophication seems limited – rather, life-cycle studies highlight the *relative potential* to cause a problem, and compost processes therefore tend to fair comparatively poorly.

Typically, enclosed compost plants now operate using biofilters. Some will employ both biofilters and ammonia scrubbers. Biofilters can consist of different materials used to reduce odours and to treat exhaust gases. Typical biofilters may consist of damp wood chippings. An interesting facet of the use of biofilters is that the biofilter itself may give rise to emissions of (different types of) VOCs. This issue has acquired some significance in the regulatory context surrounding mechanical biological treatment plants in Germany. This is because the emissions from biofilters are primarily of relatively benign volatile organic compounds such as terpenes. Despite their relatively benign nature, a focus on VOCs as an undifferentiated group of compounds led to a demand for more effective gas cleaning, typically through using regenerative thermal oxidation (RTO) systems, which eliminate VOCs without generating additional (more benign) VOCs. This discussion might appear to give rise to questions concerning how VOCs should be accounted for in an analysis of the effects given the range in the potential to cause harm that they present.[30] However, few studies assign significant emissions of VOCs to compost (rather, the issue is more important in the context of MBT plants, or plants seeking to 'compost' mixed wastes).

Biofilters vary in the efficiency with which they scrub different gases from the exhaust stream. Baky and Eriksson (2003) used the following figures for an enclosed reactor, shown in A more recent study by Amlinger and Cuhls presented data on the efficiency of CH_4 removal from biofilters as part of a wider study on emissions from composting processes. Their data – which included measurements taken from currently operating facilities - suggested much lower efficiencies of removal than those indicated in **Fout! Ongeldige bladwijzerverwijzing.**. They concluded that biofilter system could be

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

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

expected to reduce CH_4 concentrations by a maximum of 20%, with typical removal efficiencies in the order of 15%.

Table B-1.

A more recent study by Amlinger and Cuhls presented data on the efficiency of CH_4 removal from biofilters as part of a wider study on emissions from composting processes.[31] Their data – which included measurements taken from currently operating facilities - suggested much lower efficiencies of removal than those indicated in **Fout! Ongeldige bladwijzerverwijzing.**. They concluded that biofilter system could be expected to reduce CH_4 concentrations by a maximum of 20%, with typical removal efficiencies in the order of 15%.

Table B-1: Biofilter Removal Efficiencies for Different Air Emissions

Impact	Removal Efficiency
Reduction of NH_3	99%
Reduction of N_2O	90%
Reduction of CH_4	50%

B.1.2 Direct Climate Change Emissions to Air

B.1.2.1 CO_2 Emissions

Biogenic CO_2 emissions occur at the facility during the composting process itself (discussed in Section B.1.2.1.1) and continue after the compost has been applied to land (discussed in Section B.1.2.1.2). It will be seen that the proportion of carbon released at the facility is dependent on length of time the compost is allowed to mature.

B.1.2.1.1 CO_2 Emissions during the Composting Process

In their study for the Danish EPA, Baky and Eriksson assumed that the compost was well aerated during the whole process and had been allowed to fully mature.[32] Because of the life-cycle nature of the analysis, the time profile of emissions was not accorded any significance. The study also assumed that different types of organic matter were degraded in differing proportions. Heat, CO_2, and some methane (CH_4) were assumed to be released, whilst the mature compost was assumed to consist of mainly humus-like substances. Different organic substances were deemed to decompose to differing degrees in accordance with an earlier study.[33] That study reviewed the degradation of different fractions of organic carbon as presented in three other studies. The degradation rates and the rates used are shown in Table B-2.

Komilis and Ham made measurements of emissions from compost plants in the context of work on the development of life-cycle inventories in the US.[34] As in the ORWARE model used by Baky and Eriksson, they sought to relate CO_2 emissions to the nature of the input

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

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

33 U. Sonesson (1996) Modelling of the Compost and Transport Process in the ORWARE Simulation Model, Report 214, Swedish University of Agricultural Sciences (SLU), Department of Agricultural Engineering, Uppsala Sweden

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

materials. However, rather than looking at the constituent inputs in terms of the form of the organic carbon in the materials being composted, they looked at more intuitive categories, these being the relative proportions of mixed paper, garden waste and food waste. On the basis of their experiments, they estimated that CO_2 emissions, and associated percentage loss of dry matter for the organic fraction, could be calculated from knowledge of these fractions.

Table B-2: Degradation of Organic Matter in a 'Normal Compost) as % of Incoming Material

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

Notes

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

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

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

> *'"complete" degradation was reached, as measured by the CO_2 flow rate and after ensuring that this was not due to moisture limitations. […]Nitrogen was added prior to the initiation of the experiments to achieve a C/N ratio of around 30.'*[35]

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

It is interesting to assess the consistency of the approaches of Baky and Eriksson and Komilis and Ham, and to compare these with estimates of CO_2 emissions from compost used in other studies. Schleiss used figures of 266 kg/tonne and 281 kg /tonne for enclosed automated composting and open composting in covered boxes respectively.[36]

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

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

Eunomia et al used a figure of 400 kg/tonne.[37] Schleiss's figure was based upon a waste composition which was 16.2% carbon, so this amounts to a 45-47% loss of carbon.

Approach Taken in this Study

It is assumed, in the present study, that the quantity of emissions of CO_2 is not dependent upon the nature of the facility. What can be said, however, is that the mineralization of CO_2 may occur more quickly or more slowly depending upon the process type and the operating parameters.

The key issue has been to relate the emissions to the input wastes. In this study we have used the approach in which different constituent elements of carbon are taken to degrade at different rates over time. Hence, the modelling of CO_2 emissions is based upon the carbon constituents of the input materials.

Where fresh compost materials are produced (as opposed to very mature compost), CO_2 emissions are deemed to be lower in the process itself. However, the less stable material is mineralized further when applied to land, so in the round, and over an extended period of time, the CO_2 emissions for the combined process 'compost plus land application' are very similar.

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

B.1.2.1.2 CO_2 Emissions after Application of Compost to Soil

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

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

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

Three pools of organic carbon are available for microbial utilisation:

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

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

Only the first two of these pools contain carbon in readily available forms for microbial utilisation. The last pool contains carbon in a highly stable form. Some microbes can utilise this pool so depletion does occur. It can also be replenished from active and slowly decomposable fractions. It is the fact that this passive pool of carbon can be maintained or increased that leads to the idea that the passive pool can act to 'sequester' carbon in the soil. Clearly, this long turnover time does appear to imply that, for all intents and purposes (certainly for any economic analysis deploying a non-zero discount rate), this carbon is not released into the atmosphere.

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

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

Consequently, application of organic matter to soil can act to increase soil organic carbon levels, though the extent 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.

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.

The pathways for conversion of the carbon are as follows:

1. The carbon can be converted from the readily available organic matter into stable organic matter (x);
2. The carbon in the readily available organic matter can be mineralised into carbon dioxide (y); and
3. The carbon in the stable organic matter can also be mineralised into carbon dioxide (z).

We have used the following figures in the analysis: x = 25%, y = 20%, z = 1%. These figures are estimates only. Relatively little is known about the different rates of transformation, and there is some evidence that the rates may be endogenous with respect to the prevailing level of soil organic matter in the soil.[38] These figures generate profiles for carbon dioxide emissions. The external costs from these emissions, discounted over time, are then assessed and included in the modelling.

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[38] Personal comm.. F. Tittarelli.

B.1.2.2 Methane Emissions

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

> *The experiments were conducted under aerobic conditions and measurements of the headspace gaseous composition revealed that methanogenic conditions did not occur. This generally simulates the conditions in actual MSW composting facilities, since,* ***even if anaerobic conditions do develop in the center of the piles, the anaerobic gases emitted would be oxidized at the surface of the piles****.*[39]

This is a common view, but one study which has measured positive emissions of methane is that of Scheiss.[40] However, measurements by Schleiss were made close to the hotspot in the centre of the compost pile, where measurements could be expected to be higher than at the surface, if one accepts that some oxidation of methane occurs through the composting biomass (and this seems likely). Schleiss gives a figure of 5.4 kg CH_4 per tonne for fully automated composting and 11.1 kg CH_4 per tonne of waste for composting in open air covered boxes. Eunomia et al used a figure of 0.983 kg per tonne.[41] Grontmij and IVAM estimate a figure of 0.195 kg per tonne for VFG facilities in the Netherlands (based upon some measurements as well as literature review), though they note some uncertainties.[42]

A more recent study by Amlinger and Cuhls presented data on emissions of CH4 from open and closed composting facilities.[43] Their study included data from a literature review as well as new measurements taken from operating facilities. Their data suggested CH_4 emissions in the raw gas from enclosed facilities ranges of between 816-1,132 g of CH_4 per tonne of material to process. Whilst the action of the biofilter might expected to remove some of the CH_4, measurements for biofilter efficiency presented by the authors suggested that typically not more than 15% of the CH_4 was removed. The same study found that CH_4 emissions for open windrows ranged from between 49 – 604 g per tonne of waste to the facility.

Approach Taken in this Study

For enclosed facilities we assume emissions of 700 g of CH_4 per tonne of waste to facility, whilst the figure for open (window) processes is taken to be 50 g of CH_4 per tonne. These values reflect the lowest values seen in Amlinger et al (2008) and are taken to be indicative of well managed composting processes.

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

[40] Konrad Schleiss (1999) *Grüngutbewirtschaftung im Kanton Zürich aus betriebswirtschlaftlicher und ökologischer Sicht: Situationsanalyse, Szenarioanalyse, ökonomische und ökologische Bewertung sowie Synthese mit MAUT*, Dissertation ETH No 13,746, 1999

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

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

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

B.1.2.3 Nitrous Oxide Emissions

There are two principle sources of nitrous oxide emissions in composting processes:

1. Direct emissions of the gas to air from the composting process itself;
2. Additional emissions resulting from the use of biofilters in enclosed processes to reduce emissions of ammonia.

These two pathways are discussed in the section that follows.

Nitrogenous Emissions to Air

Ammonia emissions are determined by the quantity of ammonium ions, urea, and organically bound nitrogen. The pH value, temperature, ventilation, and the C/N-relation constitute other influencing factors. An increasing pH value, higher temperature, and/or better ventilation lead to greater emissions. High C/N relations cause NH_3 emissions to diminish.[44]

Nitrous oxide emissions are also determined by temperature, ventilation, nitrogen content, the C/N relation, and other factors.[45] Maximum N_2O formation rates are observed if the supply of oxygen during decomposition is insufficient. This may occur, for example, if the partial pressure of oxygen in the rotting material drops to zero due to very high rates of biological activity.[46]

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

Hellmann studied the emission of the gases CO_2, CH_4, and N_2O during the composting of domestic waste.[47] The study's results are based on sampling with gas flux chambers and the establishment of concentration using a gas chromatograph. The total CH_4-C and N_2O-N emissions were shown in relation to the dry mass of the basic material or the total CO_2-C emission. Depending on the composting conditions, nitrous oxide emissions range from 12 to 114 g N_2O-N per tonne of basic dry mass for a composting period of 89 days. In relation to the nitrogen content of the basic dry mass (1% to 1.7%), these emissions account for approximately 0.1 to 0.8% of the initial total nitrogen content. These N_2O values span a range within which falls the value proposed by Grontmij and IVAM.[48] They

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

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

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

[47] B. Hellmann (1995) *Freisetzung Klimarelevanter Spurengase in Bereichen mit hoher Akkumulation von Biomassen*, Abschlußbericht für die Deutsche Bundesstiftung Umwelt, Osnabrück, Zeller Verlag 1995

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

propose a value of 101g N_2O/tonne VFG waste, though this is not related back to N-content of the feedstock.

Ballestero and Douglas also used gas flux chambers and gas chromatography to measure the formation of nitrous oxide during the composting of manure and garden waste.[49] Within 60 days, manure composting caused 2.19% of the initial nitrogen content to be emitted as N_2O. When garden waste was composted, the percentage of N_2O emissions was considerably lower (1.18%).

Gronauer et al suggest that around 12% of total nitrogen escapes from the material in the form of ammonia.[50] This gave a figure of 0.53 kg/tonne waste in raw gas, but 0.0264 kg per tonne waste when the air is passed through a biofilter. Gronauer et al also assumed that 0.15 kg N_2O per tonne waste would be emitted.

One Swedish study[51] assumed the nitrogen leakage to air was 7.5% of the nitrogen content of the feedstock. Of this leakage, it was assumed 89% was emitted as NH_3, 9% as N_2O and 2% N_2. The study for the Danish EPA assumed that of the total amount of nitrogen lost as gaseous emission, 98 % was volatilised as NH_3, 0.5 % as N_2O and 1.5 % as N_2.[52] Clearly, these are figures for raw gas as opposed to gas which has been scrubbed.

Komilis and Ham took the view that depending on the C:N-ratio in the feedstock material, part of the nitrogen present would be volatilised as gas, primarily as NH_3, but also as N_2O and N_2. As with CO_2, they developed a basis for modelling the emissions of ammonia based upon the dry-matter proportions of food waste, mixed paper and garden waste in the input feedstock.[53] Their equation appears to give values which are much too high, especially in composts with high proportions of food waste (the figures appear too high by a factor of 10 or so).

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

Eunomia et al used figures for NH_3 and N_2O of 371 g/tonne waste and 11 g/tonne waste respectively.[54] They also carried out a 'back of the envelope' calculation for an operating plant suggesting:

> *A similar attempt at calculations concerning ammonia emissions can be attempted from plant data. One 45,000 tonne plant draws through 14,000 cubic metres of exhaust air per hour with a concentration between 20 and 40 mg per cubic metre. This equates to 545 g to 1,090 g per tonne. This is consistent with other studies but*

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

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

51 Goran Finnvenden, Jessica Johansson, Per Lind and Asa Moberg (2000) Life Cycle Assessments of Energy from Solid Waste, Forskningsgruppen for Miljostrategiska Studier, FMS 137, August 2000

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

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

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

it represents the concentration of exhaust air input to a biofilter. This would be expected to reduce ammonia emissions.

More recent data presented by Amlinger and Cuhls also showed that emissions of N_2O increased in the gas after the biofilter.[55] Measurements taken in the raw gas from enclosed processes indicated a range of N_2O emissions of between 21-29 g per tonne of waste to the process, whilst those taken after the biofilter ranged from 7–9,000 g per tonne of input. Measurements of N_2O emission from windrows ranged from 116-178 g per tonne of waste to the facility.

Approach Taken in this Study

In this study, we have modelled ammonia emissions assuming 9% of the input nitrogen is converted to ammonia, 1% is converted to N_2O, resulting in emissions of 116 g N_2O per tonne of waste treated at a windrow facility. For enclosed facilities, the operation of the biofilter is assumed to result in emissions of 360 g N_2O per tonne of garden waste (the figure for a mixed food / garden waste feedstock is slightly higher at 478 g per tonne).

A more accurate approach would probably be to vary these proportions in some relation to the C:N ratio of the input material though we could find no basis upon which to do this.

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

B.1.3 Air Quality Impacts

The literature suggests that the following pollutants may be released to air in varying quantities through composting processes:

1. Nitrogenous emissions (principally NH_3);
2. VOCs;
3. Carbon Monoxide; and
4. Bioaerosols.

The formation of nitrogenous compounds has already been discussed with reference to emissions of nitrous oxide in Section B.1.2.3. Impacts associated with the other pollutants are discussed in the sections that follow.

B.1.3.1 Emissions of VOCs

Relatively few studies make reference to emissions of VOCs. Komilis and Ham include them in the total inventory, but these appear to be related to energy use on site (which is dealt with separately below). In the UK, however, the Environment Agency did measure emissions from sites and their data is shown in Table B-3.

Table B-3: Emissions of VOCs from Monitoring of Compost Facilities

Compounds Detected	Grams per tonne of MSW
m,p Xylene [108-38-3; 106-42-3]	0.81
Nonane [111-84-2]	0.44
o Xylene [95-47-6]	0.54

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

Beta.-Pinene [127-91-3]	3.7
Ocimene [13877-91-3]	3.0
D-Limonene [5989-27-5]	10.5
Undecane [1120-21-4]	2.4
Dodecane [112-40-3]	1.2
Methyl-(methylethyl)-Cyclohexane [99-82-1]	1.5
TOTAL	*24.0*

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

Approach Taken in this Study

We have used the Environment Agency figure, and further assume that the use of biofilters reduces the emissions by 80% in the case of in-vessel facilities. The use of biofilters is assumed to result in zero damage cost for the remaining 20% of VOC emission (i.e. the biofilter is assumed to remove those pollutants that result in the health effects).

B.1.3.2 Carbon Monoxide

One study by Schleiss, gives a value for emissions of CO as 0.069 kg per tonne of waste input. It is possible that this measurement was made close to ‘hot spots’ and that any CO could be oxidised in passing through the biomass (and biofilter). We have assumed no emissions of carbon monoxide.

B.2 Energy Use

B.2.1 Energy Used at Facilities

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

The study of Finnvenden et al assumed the consumption of 54.4 MJ of electricity per tonne of food waste. It was also assumed that diesel would be consumed in the wheel loader, different types of mills and a screen, this amounting to 555.5 MJ/tonne food waste.[56] This latter figure – equivalent to more than 10 litres of diesel per tonne of waste - is enormous and cannot be considered seriously.

Baky and Eriksson used different quantities of energy depending upon whether the composting process was enclosed or in-vessel. The figures used are shown in Table B-4. It can be seen that the figures vary significantly across the processes, and that they differ considerably from those of Finnvenden et al. The diesel use for the open air facility is extremely low.

Table B-4: Energy Use at Open Air Windrows and Reactor Composts

Types of compost facility	Type of energy	Energy use (MJ / tonne)
Open windrow	Electricity	0.0

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

	Diesel	1.5
Reactor (enclosed) compost facility	Electricity	180.1
	Diesel	75.5

Source Baky A and Eriksson O (2003) Systems Analysis of Organic Waste Management in Denmark, Miljøstyrelsen, Copenhagen

Eunomia et al assumed that both diesel and electricity would be used. It was assumed that 50 kWh (180 MJ) per tonne of waste would be used and 1 litre (around 42 MJ) of diesel.[57]

A US study by Diaz et al. suggests average energy requirements in a MSW composting facility are 34.4 kW h/ t of MSW.[58] The measured values include energy consumed directly within the facility, including extensive pre-processing prior to composting (e.g., size reduction, screening). However, no odour control system was included. These figures are close to those given by Grontmij and IVAM in recent work for the Netherlands Association of Waste Management Companies. An inventory of VFG- composting facilities revealed an average value of 29 kWh per tonne of waste.[59]

A more recent US study suggested values of 97 kWh/t and 167 kWh/t for high quality (HQCF) and low-quality facilities (LQCF), respectively, with only 29 kWh/t for open-air windrow facilities.[60] These higher values are partly a reflection of the fact that pre-combustion and combustion energy requirements were included in the more recent study. It is worth reporting some of the findings in more detail:

> *Based on the HQCF, 29, 56, 3, 7, and 1% of the total energy requirements are due to hammermill, odor control, front end loaders, windrow turner, and trommel screen operation, respectively, while 4% is due to building operation. Relatively similar values apply to the LQCF. In the case of the YWCF* [garden waste composting facility]*, 55, 16, 8, and 21% of total energy are due to the tub grinder, the front-end loader, screens and building operation, respectively.*
>
> *In the LQCF and HQCF, electricity related energy accounts for more than 90% of the total energy with the rest being diesel derived energy. In the YWCF, 60% of the total energy is diesel combustion energy with 29% being electricity and the rest being diesel pre-combustion energy. This is because only diesel-powered equipment is used in the YWCF and electricity is limited to the building operation.*
>
> *[…] Due to the extensive usage of electricity in the LQCF and HQCF diesel combustion is responsible for less than 50% of* [the energy-related emissions within the boundary of the facility] *with the exception of CO. […] Diesel combustion is generally responsible for production of a relatively large percentage of the NO_x and CO emissions from both the MSW composting facilities. SO_x emissions are primarily produced due to electricity consumption and are therefore produced outside the*

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

58 L. F. Diaz, C. G. Golueke and G. M. Savage, G. M. (1986). Energetics of compost production and utilization BioCycle, 27(8), 49–54.

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

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

boundaries of the facility itself. Because of the limited use of electricity in the YWCF, the majority of the atmospheric emissions are due to diesel combustion.

The study also notes that for the HQCF:

Approximately 90% of the overall fossil CO_2 emitted from the aforementioned facility—i.e., approximately 33 kg CO_2 / t MSW—is a result of electrical usage. However, 92, 91, and 98% of the overall CO_2 emissions from the LQCF, HQCF, and YWCF, respectively, are due to the decomposition of the organic substrate. Essentially all of the ammonia emitted from all facilities is due to decomposition.

The sensitivity of energy usage to specific variables was also tested in the study (see Table B-5). The significant variation with some of the parameters makes clear why generalization is so difficult in this area. Importantly, since the HQCF was a plant treating mixed waste, the hammermill design was found to have a marked effect on total energy and emissions, since it accounts for a large usage of electrical energy. The study found that pre-sorting would affect the shredding coefficient by changing it from 1 to 0.67, resulting in reductions in total energy usage and associated gaseous environmental pollutants.

Table B-5: Sensitivity of Energy Use to Variation in Parameters

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

Source: Dimitris P. Komilis and Robert K. Ham (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, Journal of Environmental Engineering, Vol. 130, No. 11, November 1, 2004, p.1,394

A recent literature review on the greenhouse gas emissions from composting processes found that windrow facilities used between 0.4 and 6.0 litres of diesel per tonne of waste input, and suggested that 3 litres was considered typical. For enclosed facilities the study found a range of between 0.13 and 3.0 litres per tonne of waste, with the lower end of the range considered typical.[61] The same review suggested that open facilities used 0.023 – 19.7 kWh of electricity per tonne of waste with most facilities using amounts at the lower end of the range; the range for enclosed facilities was 9 – 65 kWh.

Approach Taken in this Study

We have assumed the following energy use at composting facilities, taking into account the values from the literature previously cited:

- For windrow facilities: 1 litre of diesel and 0 kWh of electricity;
- For enclosed facilities: 0.3 litres of diesel and 40 kWh of electricity.

Impacts (in terms of the external costs per kWh of energy) associated with the use of electricity at facilities will vary depending on both the energy mix of the country, and the

[61] Boldrin A, Anderson J, Moller J and Christensen T (2009) Composting and Compost Utilisation: Accounting of Greenhouse Gases and Global Warming Contribution, Waste Management and Research, Article in Press

damage cost associated with each tonne of pollutant, as was previously discussed in Section A.4.4.4.

B.2.2 Energy Used to Spread Compost

The literature review produced by Boldrin et al suggested that 0.17-0.60 litres of diesel would be required to spread the compost produced from one tonne of food waste whilst the range for compost produced from garden waste was 0.19-0.40 litre.[62] Their figures were based on the nitrogen content of the compost, assuming the fulfilment of the EU Nitrate Directive. An earlier study by Dalemo suggested diesel use for the spreading of residues of 12 litres per hectare.[63] Using Dalemo's figure in our model indicates diesel use of 0.54 litres per tonne for a mixed food / garden compost, or 0.41 litres per tonne assuming the feedstock is garden waste. We have used the figure given by Dalemo in our model.

B.3 Monetised Benefits Associated With the Use of Compost

Compost can be used in agriculture, horticulture or hobby gardening. The environmental benefits from each of these uses may be different, depending on which product the compost is assumed to displace.

Thus whilst compost used in agriculture typically displaces the use of alternative nutrient sources such as manure or synthetic fertilisers, where it is used in horticultural or amateur gardening compost usually displaces the use of peat-based soil improvers or growing media.

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

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

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

The application of stabilised compost to soil can affect soil fertility by modifying soil chemical, physical and biological properties. Compost can result in the storage of inorganic plant nutrients, affect the soil's ion exchange capacity, chelating ability and

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[62] Boldrin A, Anderson J, Moller J and Christensen T (2009) Composting and Compost Utilisation: Accounting of Greenhouse Gases and Global Warming Contribution, Waste Management and Research, Article in Press

[63] Dalemo (1997) ORWARE – A Simulation Model for Organic Waste Handling Systems, Part 1: Model Description, Resources, Conservation and Recycling, 21, pp17-37

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.

B.3.1 Quantity and Quality of Compost Output

The quantity of compost resulting from any process is likely to relate to the nature of the process, the retention time in the active phase, and the time for curing (or maturation) of the compost.

The quantity of material is derived through use of an equation for dry matter loss (from Komilis and Ham).[64] The remaining mass of material has not been calculated but is taken from empirical data concerning the quantity of compost produced from different feedstocks. We assume that 400 kg of compost is produced from one tonne of garden waste sent to an IVC or windrow composting process.

B.3.2 Uses of Compost

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.

Previous work by Eunomia et al on the use of compost in Europe collated information regarding the types of crops to which compost and digestate were applied from questionnaire responses.[65] The survey reponses indicated that in France, 71 % of the predominantly mixed waste products produced from composting and AD facilities were used in agriculture in 2005, with 68 % used in cereal production and 3 % used in

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

[65] Eunomia, J. Barth, E. Favoino and F. Amlinger (2009) *OAV024 – Frameworks for Use of Compost in Agriculture in Europe*, Report for WRAP, January 2009

specialised agriculture such as vineyards and vegetable crops. In 2003, 76 % of such products were used in agriculture, with 58 % used in cereal production, 12 % in vineyards, and 6 % in organic agriculture.

In Sweden, compost is not used in agriculture, due to the perception from farmers that compost contains less nitrogen, and because the growing media/substrate market is much more economically viable for the compost plants. 98 % of digestate is, however, used in agriculture, with almost all of the digestate used for cereal production (particularly barley and wheat).

The following graphs present data from the survey responses, indicating the proportion of compost output from different types of facilities that is used within the agricultural sector, for each of the member states included within the study. Figure 5 shows the agricultural sector's market share for all types of biowaste treatment products. Figure 6 presents the same for the output from composting facilities, whilst Figure 7 shows the same for the AD facilities. These figures indicate that the output from AD facilities is much more likely to be used within agriculture than is the case with the compost products produced from composting facilities.

On the basis of the survey responses presented in these graphics, the current analysis assumes the following:

- 50% of the compost produced from IVC and windrow facilities is used in agriculture, with the remainder used in horticultural / amateur gardening applications;
- 90% of the compost produced from AD facilities is used in agricultural applications with the remainder used in horticulture and amateur gardening.

Where compost is used in agriculture it is assumed to displace the use of synthetic fertilisers whilst that used in horticulture and amateur gardening is assumed to displace the use of peat. Monetised benefits for use of the compost in agriculture are discussed in Section B.3.3, whilst those for gardening and horticulture are discussed in Section B.3.4. All of the output from home composting is assumed to displace the use of peat in the garden.

Figure 5: Agricultural Sector Market Share – Overall Biowaste Treatment Products

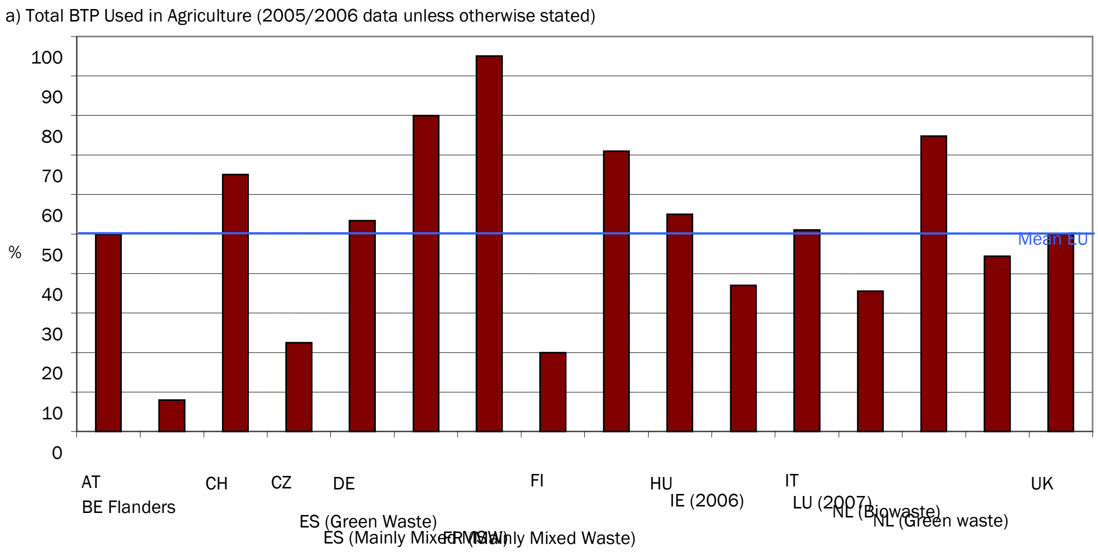


Figure 6: Agricultural Sector Market Share – Products Produced from Composting Facilities

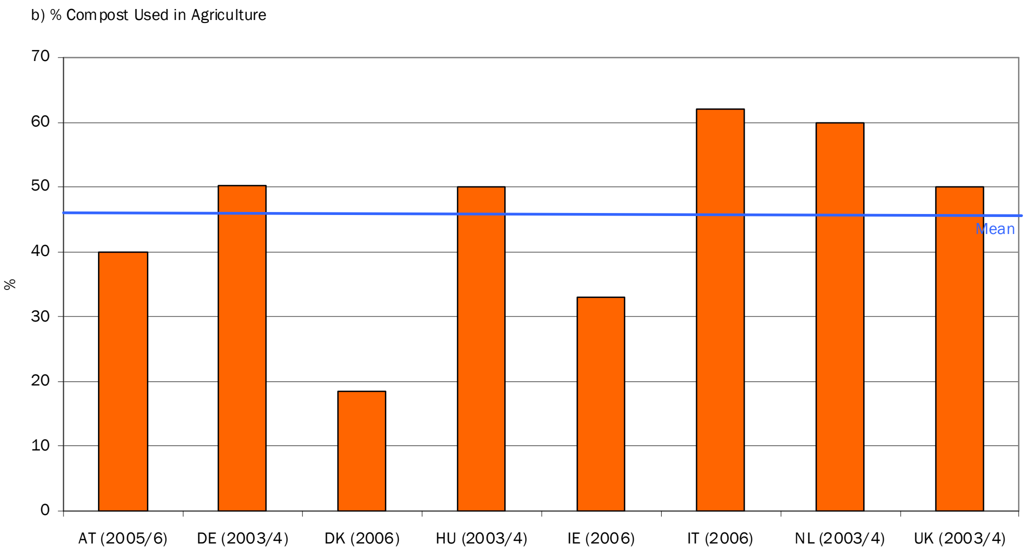
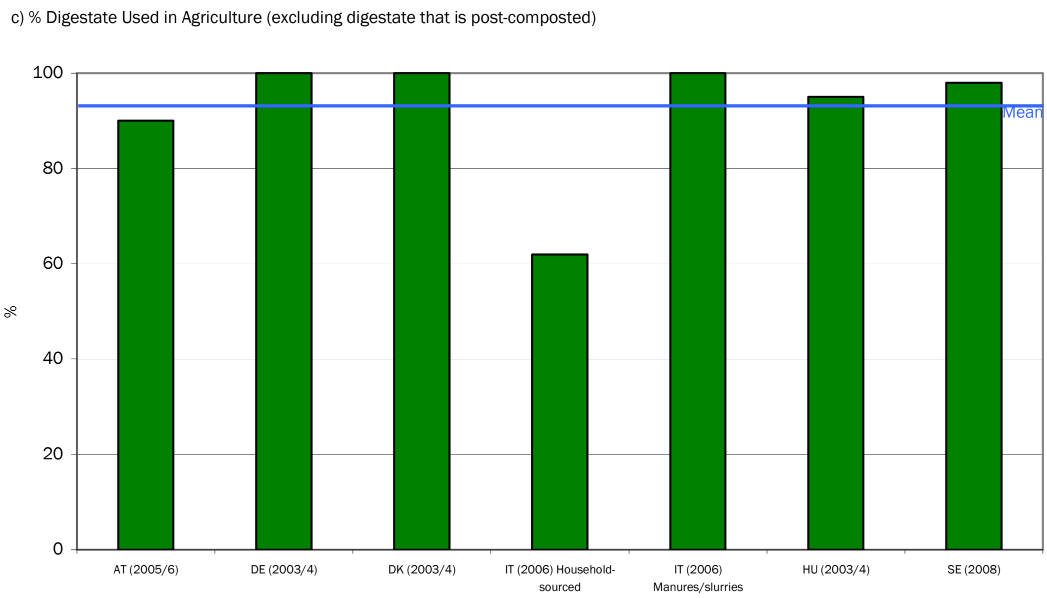


Figure 7: Agricultural Sector Market Share – Products Produced from AD Facilities



The modelling assumes the production of relatively mature compost that would not be required for most agricultural applications, though it would be essential in higher quality and value uses such as in potting mixes.

B.3.3 The Use of Compost in Agriculture

Principal benefits associated with the use of compost in agriculture are:

- A reduction in the use of fertilisers. The following environmental impacts are assumed to be offset:
  - The manufacture of fertiliser (an energy intensive process);
  - The use of fertiliser itself (principally nitrous oxide emissions to air, and the leaching of nitrate to water from soil)
- A reduction in pesticide use;
- A reduction in water use.

Evidence for these benefits is presented in the sections that follow, along with the monetisation of the impact. Impacts in terms of emissions to soil and water are considered here, as well as those resulting from emissions to air.

B.3.3.1 Compost as a Replacement for Mineral Fertiliser

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 B-6 provides a summary of some of the advantages and disadvantages of using compost compared to mineral fertilisers in agriculture. The composting industry needs to overcome the disadvantages and promote the advantages of compost if it is to convince farmers that compost is a viable alternative or can be used in conjunction with conventional fertilisers. This will effectively ‘internalise’, in market prices, some of the external benefits which we are about to explore.

Table B-6 Advantages And Disadvantages Of Using Compost Compared To Mineral Fertilisers

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

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

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

B.3.3.2 Modelling the Amount of Nutrient Displaced and the Rate of Application

In principle, it would be possible to calculate the nutrient content from a substance flow analysis. However, this was felt to be unnecessary given that more accurate empirical data exists from actual measurements of nutrient content. Readily available, real data was used in this context rather than adopting the substance flow route, with potential to give rise to errors. The nutrient values we have used for composts are shown in Table B-7.

Table B-7: Nutrient Content of Composts with Different Biowaste Components

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

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

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

The assumption of ‘one-for-one’ nutrient displacement is, however, an unrealistic one to the extent that one is implying a perfect optimisation of the replacement process. Such a situation is only likely to be achieved where farmers are well informed about the nutrient content of the matter being applied.

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.

Approach Taken in this Study

We have assumed that 10 tonnes of dry matter is applied per hectare.

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

For synthetic fertilisers, a loss rate of 23% is assumed for nitrogenous fertilisers.[66] 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 B-8 below. Equivalent projections for P and K displacement have been calculated using mineralization rates of 70% for both.

Table B-8: Evolution in N Displacement Associated with 10 tonnes Dry Matter of Composting Applied to Farmland, Southern Member State Case

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

B.3.3.3 Impacts Resulting from the Manufacture of Fertiliser

Environmental impacts resulting from the manufacture of fertiliser includes the energy required in the production process, as well as that required to produce the pre-cursor products. Our model also includes a factor for the cost to industry of clean up from phosphogypsum disposal. These are discussed in the sections that follow.

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

External Costs of Fertiliser Manufacture

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

There are a number of different routes to NPK fertiliser production. The most widespread are the mixed acid and nitrophosphate routes. This study used two sources of data, the first on Best Available Technologies from EFMA (European Fertilisers Manufacturing Association), the second from Davies and Haglund.[67] The first of these sources refers to production via the mixed acid route. There are three further distinctions within this route (granulation with a pipe reactor system, drum granulation with ammoniation and digestion). The data concerning emissions have been taken from those for the 'digestion' process because it is the only process for which this study has obtained data associated with all the required raw materials. The EFMA booklets suggest that these three processes cover the majority of NPK fertiliser production in Europe. The data from Davies and Haglund are calculated from, separately, production of ammonium nitrate, Triple Superphosphate and (for potassium), PK fertiliser (22% P_2O_5, 22% K_2O).

Avoided Phosphate Rock Extraction

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%).[68] Energy use for producing phosphate rock has been estimated at 73.5 kWh/tonne.[69] In a previous report, Eunomia simply attributed additional energy consumption for phosphate fertiliser on this basis. However, in this report, we have used the estimated damages associated with phosphate rock extraction from Nolan-ITU.[70] The figure quoted there is converted to UK sterling and adjusted to current sterling using UK GDP deflators.

Avoided Process Wastewater Disposal Phosphate Fertiliser

Soulsby et al cite 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.[71] 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

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[67] J. Davies and C. Haglund (1999) *Life Cycle Inventory (LCI) of Fertiliser Production – Fertiliser Products Used in Sweden and Western Europe*, SIKreport, No 654 1999, SIK The Swedish Institute for food and biotechnology, Gothernburg, Sweden.

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

[69] UNEP and UNIDO (United Nations Environment Programme And United Nations Industrial Development Organisation) (1998). Mineral Fertilizer Production and the Environment – Part 1: The Fertilizer Industry's Manufacturing Processes and Environmental Issues. In collaboration with the International Fertilizer Industry Association. Technical Report Number 26 – Part 1. UNEP and UNIDO, Paris and Vienna. 1998.

[70] Nolan-ITU (2004) *TBL Assessment of Garden Organics Management*, Final Report to the NSW Dept of Environment and Conservation, Sustainability Programs Division, May 2004.

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

scenarios at the 21 locations range from \$380 million (£233 million) p.a. to nearly \$1 billion (£613 billion) p.a. 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 €58.24 per tonne of, P_2O_5 (taking the lower value from the study). 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).

Modelling Approach Taken in this Study

The EFMA booklets provide gaseous emissions and energy consumption data associated with sulphuric acid, nitric acid, phosphoric acid and ammonia production, the base acids used in mixed acid production of NPK (15:15:15). The data also quotes the raw material requirement for the mixed acid route, in terms of sulphuric acid, nitric acid, phosphoric acid, ammonia and phosphate rock. Having derived the emissions and energy requirement associated with the production of each of these materials (e.g. extraction in terms of phosphate rock), these were factored according to the relative proportions used in the NPK (15:15:15) fertiliser. It was then assumed that for each of the nutrient components, that one-third of the processing requirement was attributable to the manufacture of 150 kg N, one-third to the manufacture of 150kg K_2O 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).

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

B.3.3.4 Impacts Resulting from the Use of Fertiliser

Our model considers the following impacts occurring as a result of the use of fertilisers:

1. Leaching of nitrate to water from soil;
2. N_2O emissions to air.

These are discussed in the sections that follow.

B.3.3.4.1 Nitrate Contamination from Fertiliser Use

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 the 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 via inorganic nitrogen fertilisers also has the potential to volatise resulting in the loss of more than 50% of the 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 the maximum N application rates in the Code of Good Agricultural Practise without significantly increasing nitrate leaching. This should be related to the high percentage of slow-release N (organic N) that tends to be released over longer time frames than in the case of chemical fertilisation. [72]

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.

External Costs of Nitrate Pollution of Water Supplies

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.

Some studies have used the shadow price of nitrogen to attribute the impact of nitrate pollution. In a report for the Croatian Government detailing the external costs of different farming methods, Pretty et al presented data on the shadow price from a range of European studies carried out over the previous decades.[73] This data is presented in Table B-9.

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

[73] *Znaor D, Pretty J, Morrison, J and Todorovic S K (2005) Environmental and Macroeconomic Impact Assessment of Different Development Scenarios to Organic and Low-input Farming in Croatia, Report for the Republic Government of Croatia published by University of Essex*

Table B-9: Estimates for the Shadow Price of Nitrogen for European Countries

Country	**Authors**	**Reference year**	**Average nitrogen shadow price €**	
			Ref year	2005
The Netherlands	Bleijenberg, Davidson et al 1998	1997	1.6	2.4
	Reinhard 1999	1998	1.4	2.0
	Davidson, Hof et al 2002	2002	3.8	4.4
Italy	Tiezzi 1999	1991	0.3	0.6
Germany	Piot-Lepetit, Brummer et al, 2002	1998	2.5	3.5
Denmark	Schou, Skop et al 2000	1999	1.6	2.1
	Berentsen, Giesen et al 1998	2002	4.5	5.2
Norway	Vatn, Bakken et al 1999	1997	1.8	2.7
Sweden	Bystrom 1998	1997	3.0	4.4
Finland	Lankoski and Ollikainen 1999	1998	5.4	7.6
France	Piot-Lepetit and Vermersch 1998	1997	1.8	2.7
	Piot-Lepetit, Brummer et al 2002	1998	1.4	1.9
EU-15	Brouwer, Hellegers et al 1999	1999	2.0	2.6
USA	Shaik, Helmers et al 2002	1997	2.4	3.6
Danube Basin	Wit, Posma et al 1999	1997	1.0	1.5
	Gren, Broth et al 1995	1994	2.2	3.7
Croatia	Sumelius, Mesic et al 2005	2002	1.4	1.6
	Average		*2.2*	*3.1*

Cost of Nitrate Removal from Water Supplies

The cost of removing nitrate from groundwater is not insignificant. A study in the UK estimated the cost of nitrate removal at £18.8 million per year in the years 1992-1997 (capital expenditure) plus £1.7 million (operating expenditure.[74] The report 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 50 mg/l.[75] Grossing up on the basis of 835,212 households,[76] 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 carried out in Gotland in Sweden sought to elicit preferences for nitrate pollution in aquifers to be

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

[75] N. Hanley (1990) The Economics of Nitrate Pollution, *European Review of Agricultural Economics*, 17, pp.129-51.

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

reduced to levels below the WHO recommended limit. This value was SEK 600 (1995 prices) per person per year.[77]

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

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

Modelling Approach Taken in this Study

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 B-8. 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 **Fout! Verwijzingsbron niet gevonden.**

Again, given the limited availability of such values, we have used a figure of €3.20 per kg N, based on the average shadow price of nitrogen quoted by Pretty et al (see Table B-9).

These enable the development of estimates as to the value of the 'avoided leaching' associated with the nitrogen displacement.

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

B.3.3.4.2 Impact of Nitrogenous Fertilisers on Soil N_2O Emissions

Nitrous oxide emissions from soil are complex since the gas is simultaneously produced and consumed in soils through processes of de-nitrification, 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_2O. 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.[78] 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.[79] A Dutch study cites figures for N_2O losses as between 1 and 3% of mineral N applied.[80]

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[77] I.-M. Gren, H. H. Groth, and M. Sylvén (1995): Economic Values of the Danube Floodplains. *Journal of Environmental Management* 45:333-345.

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

[79] I. P. McTaggart, B. C. Ball, and C. A. Watson (1998) *Influence on Land Use in Scotland on Soil N2O Emissions and Atmospheric CH_4 Uptake*, British Society of Soil Science Annual Meeting, Queen's University, Belfast, September 1998.

[80] A. R. Mosier (1993) Nitrous Oxide Emissions from Agricultural Soils, Proceedings of Methane and Nitrous Oxide - Methods of National Emissions Inventories and Options for Control, RIVM, Netherlands.

The following nitrogen volatilisation rates have been derived using the MANNER Model[81]. The application of slurry (6% dry matter) onto grassland in the spring at a rate to supply 250 kg of fresh nitrogen, results in 46 kg 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 250 kg of nitrogen, the nitrogen volatilisation is approximately 32 kg/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).[82]

Valuation of External Costs

The valuation of reduced N_2O emissions from fertiliser applications clearly depends upon:

- The rate at which one assumes compost replaces nitrate fertilisers and / or other products (which, as discussed above, is not known with certainty, and is likely to vary with knowledge of the product and familiarity with its use); and
- The relative rates of N_2O 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.

Modelling Approach Taken in this Study

In the analysis, we assume that, relative to compost, 0.5% of nitrogen applied as fertiliser is lost as N_2O. This is combined with the time profile of N replacement for the compost as outlined in earlier sections. Hence, the externality also depends upon the mineralisation rate.

B.3.3.5 Reduced Pesticide Use

Use of Compost to Aid Disease Suppression

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

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

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

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

[81] 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.
[82] Figure calculated using data from HDRA Consultants Compost Analysis Database which contains data from over 80 different composts.

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.[83] In addition, several studies have shown soil microbial activity to be higher, and disease incidence lower, after the application of compost. Table B-10 gives a summary of specific work carried out on specific plant pathogens. In these studies, various degrees of control were achieved by using compost.

Table B-10: 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) Basis for the Control of Soil Borne Plant Pathogens with Composts, Annual Review of Phytopathology ***24****, 9; Ringer, C. E., Millner, P.D., Teerlinck, L.M. and Lyman, B.W. (1997) Suppression of Seedling Damping-off Disease in Potting Mix Containing Animal Manure Composts. Compost Science & Utilization,* ***5****, 6-14; Yohalem, D. S., R. F. Harris, and J. H. Andrews (1994) Aqueous Extracts of Spent Mushroom Substrate for Foliar Disease Control. Compost Science & Utilization,* ***2****, 67-74*

External Costs of Pesticide Use

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

A number of attempts have been made to estimate in monetary terms the environmental costs of using pesticides.[84] Although much of the literature estimating the external costs of pesticide dates from the 1990s, a major European project looking at empirical measurement of the external costs of pesticides across the EU commenced in 2008 and is due to finish in 2011.[85]

One study estimated the total external costs of pesticide use in US agriculture at just under $8 billion against private expenditure on pesticides of some $4 billion.[86] 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.[87] They estimate benefits associated with reduced crop losses from using pesticides of some $16 billion.

In another US study, the external costs are estimated at the much lower level of between $1.3 and $3.6 billion. This results from omitting the estimates for pesticide resistance and crop losses. It should be noted, however, that the range of estimates associated with loss of biodiversity range from $0.3 to $20 billion.[88]

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.[89]

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.[90] Globally, this study suggests a figure of $4 billion, apparently on the basis of WHO estimates of 40,000 unintentional deaths per year resulting from pesticides. They also point out that biodiversity losses are potentially the least well understood of the external costs. Not

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

[85] European Commission (2009) Strengthening the Knowledge-Based Bio-Economy – The EU Support to (Plant Health and Protection) Research: On-going Activities and Roadmap for the Future

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

[87] 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).

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

[89] See Barbara Dinham (1998), The Costs of Pesticides, *Pesticides News*, No.39, March 1998, p.4.).

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

untypically, at the higher end of the range, they can swamp the estimated total external costs of pesticides.[91]

A more recent UK study by Pretty et al, which the authors believe give estimates of pesticide externalities on a conservative basis, estimates these to be (in the UK situation) of the order £8.6 per kg active ingredient used.[92] Another study used a contingent valuation survey to estimate an externality per kg of active ingredient of £12 per kg on average.[93] 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).

Modelling Approach Taken in this Study

In this study, we assume that where compost is applied at 10 tonnes dry matter per hectare, the use of pesticides falls by 20%. This may be a conservative assumption. A US study gives an example of a farm where 50% reductions in synthetic pesticides and fertilisers were achieved.[94] 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.[95] 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).

We assume that external costs of pesticide use are €13 per kg of active ingredient used, this figure being based on the external costs of pesticide use given in the two studies cited above. Figures from the Pesticide Usage Survey indicate that the UK used around 31 million kg of active ingredient in 2004.[96] This is applied over a land area of around 5 million hectares, an average of just over 6 kg active ingredient per hectare.

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

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[91] 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.).

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

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

[94] Edward C. Jaenicke (1998) *From the Ground Up: Exploring Soil Quality's Contribution to Environmental Health*, mimeo from the University of Tennessee, October 1998.

[95] A. Meier-Ploeger, H. Vogtmann and H Zehrm (1996) *Eco Balance Compost Versus NPK Fertiliser*, INAC for ORCA, Brussels: ORCA.

[96] Pesticide Usage Statistics http://pusstats.csl.gov.uk/mygraphresults.cfm.

be weighted towards sectors of agriculture which make greater use of pesticides, such as horticulture and fruit and vegetable growing.

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

B.3.3.6 Avoided Water Use

Water Stress in Member States

A recent report by the European Environment Agency noted that whilst water was relatively abundant across the continent, demand exceeded availability in many locations.[97] The same study suggested the cost to Europe of droughts occurring during the past 30 years was €100 billion.

Water use in individual Member States was examined in an earlier project carried out by the University of Kassel on behalf of the European Environment Agency, using data from Eurostat. Output from the project included a series of maps identifying areas of water stress across Europe. The water stress indicators were calculated using a Water Exploitation Index, calculated on the basis of the ratio of water withdrawal to water availability, using data obtained for the year 2000. This data is summarised in Table B-11.

The data shows that the majority of Member States were considered to have low levels of water stress in the year 2000. Those countries with significant areas of severe water shortages (defined as a water exploitation index of greater than 40%) were Belgium, Italy, the Netherlands and Spain.

[97] European Environment Agency (2009) Water Resources – Confronting Water Scarcity and Drought

Table B-11: Water Stress in Member States

	Water stress level
Austria	Low / Medium
Belgium	High
Bulgaria	Low
Cyprus	Low
Czech Republic	Low / Medium
Denmark	Low
Estonia	Low
Finland	Low
France	Low
Germany	Medium
Greece	No data
Hungary	Low
Ireland	Low
Italy	Medium / High
Latvia	Low
Lithuania	Low
Luxembourg	Low
Malta	Low
Netherlands	Medium / High
Poland	Low / Medium
Portugal	Low / Medium
Romania	Low
Slovak Republic	Low
Slovenia	Low
Spain	Medium / High
Sweden	Low
UK	Low / Medium
Notes Water stress categories based on the water exploitation index (withdrawal to availability ratio), expressed as a percentage. Categories are as follows: Low: 0-20% Medium: 20-40% High: more than 40%	

Source: Center for Environmental Systems Research (2005) Water Stress in Europe, for the European Environment Agency, available from http://dataservice.eea.europa.eu/atlas/viewdata/viewpub.asp?id=1718

The Impact of Compost Application on Soil Water Retention

Studies have indicated that the application of composted products can enhance the water use efficiency by improving infiltration and storage in the root zone and reducing deep drainage, run-off, and evaporation, and water use by weeds. The beneficial effects of compost application arise from improvements in soil physical and chemical properties.[98]

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98 A. Shiralipour, D. B. McConnel and W. H. Smith (1992) Physical and Chemical Properties of Soils as Affected by Municipal Solid Waste Compost Application, Biomass and Bioenergy 3(3-4): 261-266; S.A.R. Movahedi Naeini and H. F. Cook (2000) Influence of Municipal Compost on Temperature, Water, Nutrient Status and the Yield of Maize in a Temperate Soil. Soil Use and Management 16:215-221; L. M Bresson, C. Koch, Y. Le Bissonnais, E. Barriuso and V. Lecomte (2001) Soil Surface Structure Stabilization by Municipal

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

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

Use of composted mulch in cropping lands can also significantly increase the water use efficiency by lowering the evaporation losses from soil surface. Mulching reduces radiation and wind speed at the surface and hence, reduces the water evaporation from soil surface. Researchers have reported that surface application of mulch has resulted in reduction of between 30–70% of irrigation water required by crops due to the reduction of water evaporation from soil surface.[99] Buckerfield and Webster showed that, in South Australia, the surface application of organic mulches to vineyard soils resulted in a 34% increase in soil moisture content and an increase in grape yield.[100] This reduced evaporation offers obvious benefits for irrigated agriculture.

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

On the basis of a literature review, one study has sought to plot relationships between compost applications as soil improvers (Figure 8) and mulches (Figure 9) on soil moisture content in the 0-15 cm layer.[101] This was used to estimate the quantity of water saved

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Waste Compost Application. Soil Sci. Soc. Am. J. 65:1804-1811; J. Albaladejo, V. Castillo and E. Diaz (2000) Soil Loss and Runoff on Semiarid Land as Amended with Urban Solid Refuse, Land Degradation & Development 11: 363-373; M. Agassi, A. Hadas, Y. Benyamini, G. J. Levy, L. Kautsky, L. Avrahamov and H. Zhevelev (1998) Mulching Effect of Composted MSW on Water Percolation and Compost Degradation Rate. Comp. Sci. Util. 6(3): 34-41

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

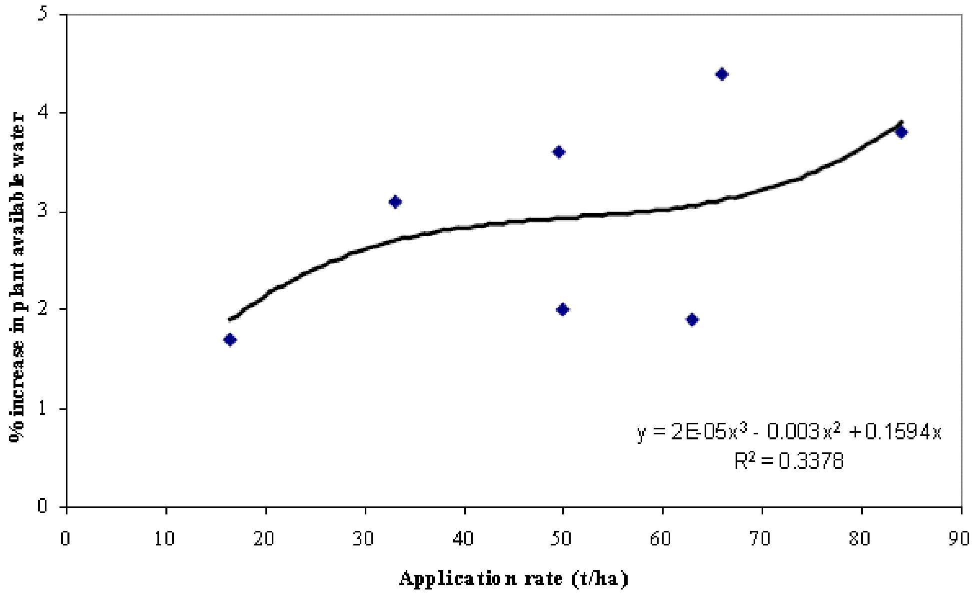
100 J. C. Buckerfield and K. A. Webster (1995) Earthworms, mulching, soil moisture and grape yields: earthworm response to soil management practices in vineyards, Barossa Valley, South Australia. Australian and New Zealand Wine Industry J. 11:47-53

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

through the application of compost. If the same relationships are assumed (and clearly, they may not hold across all member states), then it could be assumed that an application rate of 20 tonnes per hectare fresh matter would deliver a 2% increase in plant available water.

A study undertaken in the UK by Defra suggested that where irrigation is applied, it was applied at a rate of 131,300,000 m^3 over an area of 147,270 ha. In dry years, the figures are estimated at 439,470,000 m^3 over 282,960 ha.[102] An average of these figures gives an estimated 1,222 m^3 of water used per hectare. The Defra study assumed that the application of compost reduces this figure by 2%, or 24 m^3. However rain water was assumed to supply some of the plant available water. If this is assumed to be 60% of the total, then the 2% increase in availability implies a reduced demand of 61 m^3 of water (corresponding to a reduced water requirement of 5%).

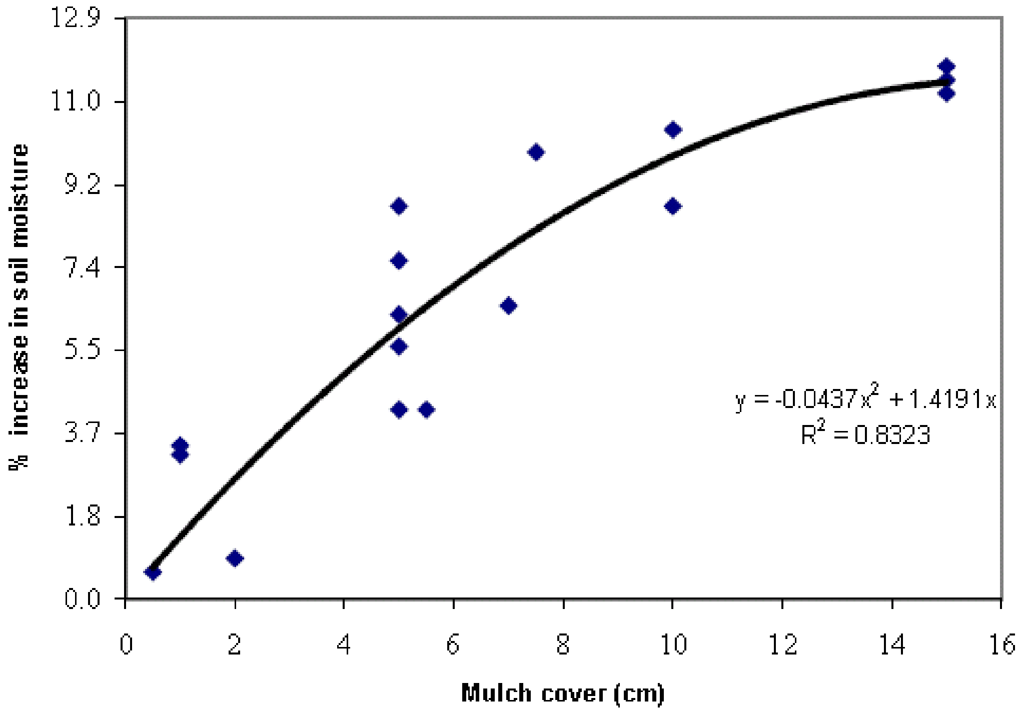
Figure 8: Effect of Compost Used as Soil Conditioner (0-15 cm layer)



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

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

Figure 9: Effect of Compost Used as Mulch on Soil Moisture (0-15 cm layer)



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

The Value of Water

The value of these savings is very difficult to estimate. The value of water is multi-dimensional. It extends beyond water *prices*, and includes values of water in terms of its 'use' and its 'non-use' values.[103] Many studies relate to the value of recreation, or to the value of wetlands, or to some measure of water quality. Avoiding abstraction can, in theory, contribute to each of these.

The EU Water Framework Directive was adopted in 2000.[104] Consistent with the polluter and user pays principles, the Directive suggested that users should pay the full costs (including the environmental as well as the resource costs) of extractive or discharge water use. Interpretation of the Directive, however, varied considerably between member states, and early guidelines were confusing.[105] There is, as yet, no European analysis linking the valuation of water with the level of water stress for each country. Member States are however required to produce and document estimates of the environmental costs of water abstraction as part of the Directive. These reports are due to be produced by each country later this year.

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

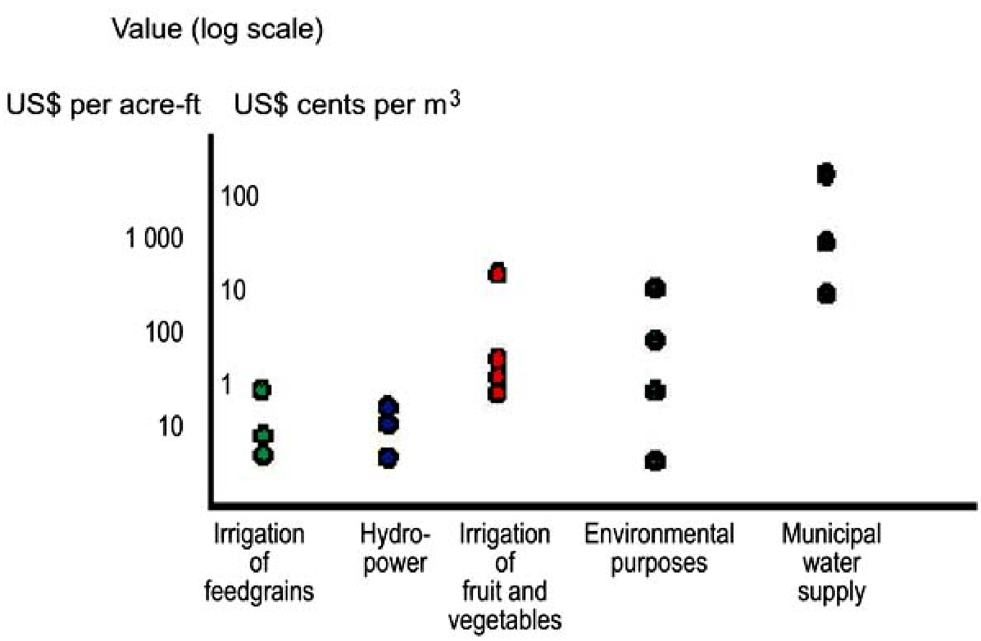
[104] European Commission (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 Establishing a Framework for Community Action in the Field of Water Policy

[105] D. Moran and S. Dann (2007) The Economic Value of Water Use: Implications for Implementing the Water Framework Directive in Scotland, *Journal of Environmental Management*, 87, pp484-496

A study in the western US reviewed a range of studies and gave figures as shown in Figure 10.[106] It suggested (amongst other things) that:

- irrigated agriculture accounts for a large proportion of water use, especially in many water-scarce areas, and the value of water for many low-value crops, such as food grains and fodder, is very low. Nevertheless, the value of water can be high (of the same order of magnitude as values in M&I end uses) where reliable supplies are used on high-value crops;
- The value of water for household purposes is usually much higher than the value for most irrigated crops. However, within household usages, the value for 'basic human needs' is much higher than the value for discretionary uses (such as garden watering). Reliability of supply is an important factor in how the resource is valued.
- The value of water for industrial purposes is typically of a similar order of magnitude to that of supplies for household purposes;
- The value of environmental and ecological purposes varies widely but typically falls between the agricultural and municipal values (as shown in Figure 10).

Figure 10: Typical Market and Non Market Values for Water Use in the Western United States of America



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

Briscoe's figures from the US suggest that a value of US$0.10 per m^3 might be a conservative estimate of the total economic value of water in the US context. The value of US$0.10 per m^3 in 1995 translates into a value of €1.07 per m^3 at current prices.

More recent analysis undertaken in Scotland in connection with implementation of the Water Framework Directive estimated the value of water used in agricultural irrigation

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

using data obtained from England and Scotland.[107] The study looked at the improvement in yields occurring as a result of the increased irrigation of potato crops and made comparisons between un-irrigated and fully irrigated crops. The value of water in the study was given as between £0.23 - £1.38 per m^3 depending upon the variety of plant, equating to €0.34 - €1.83 at current prices.

Approach Taken in this Study

In this study, we have assumed a reduced water requirement of 5% as a result of the application of compost and have used the total economic value of water provided in the US study, which is towards the upper end of the range of values given in the later Scottish study. We feel this is too important a benefit to be assigned zero value, yet there is no reasonable basis upon which to base European estimates at present. This is a major source of uncertainty in this study.

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

B.3.4 The Use of Compost in Horticulture and Amateur Gardening

The use of compost in horticulture and amateur gardening is assumed to avoid pesticide use, as was previously discussed with reference to the use of compost in agriculture (see Section B.3.3.5). Benefits are also attributed to the avoidance of peat extraction. These benefits are discussed in the following sections.

B.3.4.1 Avoided Pesticide Use

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

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

B.3.4.2 Avoided Peat Extraction

Peat Use as Soil Improver

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

[107] D. Moran and S. Dann (2007) The Economic Value of Water Use: Implications for Implementing the Water Framework Directive in Scotland, *Journal of Environmental Management*, 87, pp484-496

public as an alternative to peat. However there are several barriers to overcome before this is the case.

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

Peat Use in Growing Media

The other main use of peat is as the main constituent in growing media. Peat as opposed to compost is the preferred growing media as it has no nutritional content and has superior water holding capacity. Some research has been carried out into the use of compost as a constituent of growing media and successful results have been achieved. However, growers are reticent to use compost growing media alternatives as it is believed that they do not perform as well as peat. There is also the issue of quality, especially with regard to the presence of weed seeds and plant pathogens. Growers need to be assured that compost is free from contaminants and that it will not have any phyto-toxic effects. These assurances need to be given based on substantiated research and adequate quality and user guidelines.

Currently, it is unlikely that compost will be accepted as a blanket alternative to peat in growing media. However, there is considerable scope for blending compost with peat. As consumer confidence increases in the peat/compost mix, so could the proportion of compost in the blend, subject to the blend being fit for purpose.

Environmental Costs of Peat Extraction

Environmental costs of peat extraction include:

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

Of these impacts only external costs associated with the first two are considered, due to the inherent difficulties in attaching economic value to the others. The non-monetised impacts of peat extraction are discussed further in Section B.4.

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).[108] Unperturbed peatlands, whilst they may act as a sink for carbon, typically also emit CH_4. The balance between the two types of emission differs between the different types of peatland, with those areas that have a higher water table (such as fens) emitting more CH_4 than peat bogs.[109] However, as long

[108] 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).

[109] Cleary J, Roulet N T and Moore T R (2005) Greenhouse Gas Emissions from Canadian Peat Extraction, 1990-2000: A Life-cycle Analysis, Ambio, 34(6) pp456-461

as the peatland is unperturbed, the areas most likely retain a balance between methane emissions and carbon sequestration.

Drainage and degradation of peatlands increases carbon dioxide emissions. Some studies suggest the drainage also increases nitrous oxide emissions significantly.[110] 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.[111]

Most analyses estimating the greenhouse gas emissions associated with the use of peat relate to the use of the material as a fuel. In this case the majority of the emissions are associated with the immediate release of carbon through combustion. However a recent Canadian study looked at the greenhouse gas emissions associated with peat extraction specifically where the peat was used as a soil improver.[109] Their analysis concluded that peat decomposition associated with its end use was the largest source of greenhouse gas emissions, comprising 71% of total emissions during the 11 year period over which the study was carried out. Emissions resulting from the change in land use resulting in a switch of the peatlands from an emissions sink to a source of emissions contributed to a further 15% of the total. 10% of total emissions related to transportation whilst the extraction and processing of the peat were associated with more minor impacts.

When considering the emissions from the decompostion of recently extracted peat, the study noted that:

> *There are few empirical studies of peat decomposition that last more than 1 year. Aerobic decomposition rates of peat in the first year range from 0% to 6% for moderately to well-humidified peat. A few studies have shown rates greater than 10% in the first year, and these have generally examined the decomposition of peat mixed with other substances such as fertilizer and soil. The vast majority of extracted peat is mixed with other substances when used in horticulture, increasing peat decomposition rates. Unfortunately, it is difficult to isolate the decomposition of peat from the decomposition of the substances with which it is mixed. We used an exponential model with an annual decay rate of 5% to estimate the emission of CO_2 from decomposing peat moss.*

Approach Taken in this Study

We have adopted the annual 5% decay rate used in the previously cited Canadian study. Emissions associated with the extraction and utilisation of peat are estimated on the basis of this decomposition alone. We have not sought to model the behaviour of the peatland in-situ either prior to or immediately after the removal of the peat, as the literature suggests this behaviour will vary between different types of peatland. In this, our

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[110] N. Roulet, R. Ash, W. Quinton and T. Moore (1993) Methane Flux from Drained Northern Peatlands: Effect of a Persistent Water Table Lowering on Flux, Global Biogeochemical Cycles 7 (4) pp.749-69; K. Regina, H. Nykaken, M. Maljanen, J. Silvola and P. Martikainen (1998) Emissions of N2O and NO and Net Nitrogen Mineralization in a Boreal Forested Peatland Treated With Different Nitrogen Compounds, Canadian Journal of Forest Research, 28, pp. 132-40; C. Freeman, M. A. Lock and B. Reynolds (1992) Fluxes of CO2, CH4 and N2O from a Welsh Peatland Following Simulation of Water table Draw-down: Potential Feedback to Climate Change, Biogeochemistry, Vol.19 No.1, pp.51-60.

[111] See, for example, Environment Canada (1998) Wetland Conservation: An Excerpt from the Sinks Table Options Paper, November 1998.

approach is the same as a previously published study produced for the European Commission by AEA.[112]

The gaseous emissions associated with peat decompostion 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 200 kg/m^3. The density of compost, on the other hand, is of the order 500 kg/m^3 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.

B.4 Non Monetised Impacts

A number of environmental impacts associated with both the composting process, and the utilisation of compost have not been monetised. These impacts are discussed in the sections that follow.

B.4.1 The Production of Leachate from Composting

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

Komilis and Ham note, in their review, that varying amounts of leachate have reportedly been produced in MSW and garden waste composting facilities starting from 0 to approximately 490 litres / tonne. [113] They took the view that:

> *Given the limitations in the available data, this aspect of leaching emissions from MSW and yard wastes during and after composting was not included in this LCI. The LCI includes, however, all waterborne emissions associated with diesel and electricity pre-combustion and combustion processes.*[114]

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

Assigning an external cost – an impact given a monetary value – is extremely difficult where leachate is concerned. Many studies, even those covering landfills, where controls on leachate are, potentially, more problematic, have assigned no external cost to leachate.[115] In the case of compost facilities, given the potential problems associated with

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[112] AEA (2001) Waste Management Options and Climate Change, report for the European Commission, July 2001

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

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

115 Typically, the argument is made that the leachate is collected and tankered off for treatment, so that the treatment cost is already internalised in the operation of the landfill. This is not strictly true since a) the treatment process itself has some environmental impacts; b) the capture of leachate is not necessarily 100%, as measured instantaneously in the period of operation of the landfill; and c) where landfills are concerned, the

maintaining the biomass in a moist condition, the recirculation of leachate may be beneficial. In addition, the recirculation can reduce any issues associated with nitrogen in the leachate (though it might increase the rate of volatilisation of ammonia), and may make this available to the microbes giving rise to the degradation of material.

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

B.4.2 Odour

B.4.2.1 Causes of Odour at Composting Facilities

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

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

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

- Processing incoming feedstocks as soon as possible
- Managing the process properly, including:
    - ensure proper stabilisation of the biomass within the retention time in enclosed buildings, so as to ensure only odourless materials are present in the open curing stage
    - avoid an early refining step to reduce particle size too far, which would hinder the diffusion of air through the material that still has to complete its biochemical transformation (smaller particle size would cause the compost to lose structure and make anaerobic decomposition more likely);
- Ensuring good housekeeping practices are followed, such as:
    - Preventing the formation of leakage puddles (e.g. ensuring proper slopes to paved surfaces)
    - Avoiding external stockpiling of coarse rejects from pre-process screening steps, as these would also contain a certain percentage of fermentable materials

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

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instantaneous rate of capture of leachate in the short-term may mask much lower rates later in the life of the landfill.

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

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

B.4.2.2 Odour Treatment Using Biofilters

Biofilters are used primarily to treat odours.[116] Biofilters 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 circuiting of air (this problem is one of major maintenance issues regarding biofilters; materials with a lack of structure can shrink in as little as a few weeks, while coarse materials can last up to 5 years or so).

Biofilter media can vary from locally available compost to specifically designed media mixtures containing ingredients such as: compost, soil, peat, bark, wood, lime (deemed to withstand the tendency to acidification related to the nitrification of ammonia), lime and polystyrene spheres.[117] The desired qualities of a biofilter are outlined in Table B-12.

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[116] 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.

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

The removal efficiency of a biofilter is determined by the gas residence time in the media bed (residence time is calculated by dividing the seconds in a hour – 3600 - by the specific loading rate, expressed in $cm_{air}/hr.cm_{biofilter}$). 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%).[118]

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

Table B-12: 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

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

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

B.4.3 Other Compost Related Problems

B.4.3.1 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.

B.4.3.2 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.

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) 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.

B.4.3.3 Birds

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

B.4.3.4 Rats

It is very difficult to find a location where rats are not present at a composting facility, especially if the plant is situated in a rural location. 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.

B.4.4 Bioaerosols

Significant recent interest, at least within the UK, has centred on the potential consequences of emissions of bioaerosols from composting processes. The Environment Agency sponsored two studies in the early part of the decade. These studies were reviewed for the Cabinet Office in a paper by Hogg.[119] That review cast some doubt upon the conclusions drawn from the research undertaken, notably, the way in which it was used to influence regulatory guidance, for example, in respect of distancing from compost facilities.[120]

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 temperatures of up to 65°C. As part of its lifecycle, *Aspergillus fumigatus* produces tiny spores. If inhaled as a bioaerosol these spores may cause allergies and inflammation, which in certain individuals can cause serious health disorders such as Asthma, 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 – may be exposed to, and inhale large quantities of bioaerosols, particularly when compost is being moved or agitated. To most individuals, exposure to bioaerosols does not appear to cause significant problems. However, as with some more conventional pollutants, 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 them. In the composting process, the levels encountered can be significantly higher than background levels. 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.[121] This includes:

- 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:

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[119] D. Hogg (2002) *Waste Treatments Mk II: Health Effects*, Report for the Strategy Unit.

[120] See Environment Agency (2001) *Agency Position on Composting and Health Effects*, 2001.

[121] This is echoed in recent documentation produced by the European Agency for Safety and Health at Work. See, for example: European Agency for Safety and Health at Work (2007) European Risk Observation Report: Expert Forecast on Emerging Biological Risks related to Occupational Safety and Health, Belgium,

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

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

Generally, the problem appears to be greater at open air windrow facilities. This perspective has, for the time being, been reinforced by the Environment Agency rules concerning distances from compost facilities to dwellings. If this approach achieves its goal – that effectively, no one is exposed to levels of bioaerosols above a specific level, then it seems reasonable to suggest that the external costs associated with these emissions are unlikely to be significant other than for operatives. The review by Hogg also noted that the standard set by the Environment Agency would have implied – even in cases which were examined in the case studies – that the compost plants had to act as sinks for bioaerosols.[122]

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

The results (see Table B-13) suggest low levels of emission. Although a far more critical appraisal of these results is required, there is at least a suggestion that facilities designed to reduce the degree of turning / forced aeration can actually lead to reduced bioaerosol emissions.

[122] D. Hogg (2002) *Waste Treatments Mk II: Health Effects*, Report for the Strategy Unit.

Table B-13: Concentrations of Micro-organisms

		Moulds count per m^3	Total Viable Count per m^3
Process Not in Operation			
Inside Building	by mixer	367	33
Inside Building	on high gantry	33	17
Inside Building	by bagging unit	117	200
Inside Building	by auger extractor	83	267
Outside Building	10m downwind	17	350
Outside Building	10m downwind	133	67
Outside Building	10m downwind	117	633
Outside Building	10m upwind	33	33
Outside Building	10m upwind	50	33
Outside Building	100m downwind	33	50
Outside Building	100m downwind	0	150
Outside Building	100m downwind	17	267
Process in Operation			
Inside Building	adjacent to bagging	133	233
Outside Building	10m downwind during bagging	150	2200
Inside Building	adjacent to mixer	1267	2133
Inside Building	on gantry during top-up	467	600
Inside Building	during compost extraction	100	467
Inside Building	during compost extraction	500	3067
Inside Building	by Auger top end (RHS)	533	2533
Inside Building	by Auger bottom end (RHS)	867	1233
Inside Building	by Auger top end (IHS)	367	2750
Inside Building	by Auger bottom end (IHS)	133	833
Inside Building	by site office	333	1350
Inside Building	by bagging line (not operating)	267	300
Outside Building	in delivery bay during drop off of material	400	967
Inside Building	moving raw material to storage area	2200	2000
Outside Building	10m downwind	67	167
Outside Building	10m downwind	233	400
Outside Building	10m downwind	283	433
Outside Building	10m downwind	367	367
Outside Building	10m downwind	367	300
Outside Building	10m downwind	50	400
Outside Building	10m upwind	150	133
Outside Building	10m upwind	133	600
Outside Building	10m upwind	100	267

A more recent study undertaken at a Spanish composting facility yielded similar results to those presented in Table B-14.[123]

A review of the literature on this subject undertaken by the European Agency for Safety and Health at Work found that workers in the composting industry had an elevated risk of ill-health through the occupational exposure to biological agents, although the study suggested it was very difficult to assess the magnitude of this additional risk.[124] However, the review also suggested that employees at landfill and incineration facilities were similarly exposed.[125]

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

Approach Taken in this Study

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

B.4.5 Composting and Other Human Pathogens

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

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

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[123] Nadal M, Inza I, Schuhmacher M, Figueras M and Domingo J (2009) Health Risks of the Occupational Exposure to Microbiological and Chemical Pollutants in a Municipal Waste Organic Fraction Treatment Plant, International Journal of Hygiene and Environmental Health, 212, pp661-669

[124] European Agency for Safety and Health at Work (2007) European Risk Observation Report: Expert Forecast on Emerging Biological Risks related to Occupational Safety and Health, Belgium

[125] In the case of those employed at incineration facilities, a further study noted that additional risks were associated with the exposure of employees to inorganic chemicals. See: European Agency for Safety and Health at Work (2009) European Risk Observation Report: Expert Forecast on Emerging Chemical Risks related to Occupational Safety and Health, Belgium

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.[126] 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. One study reported that *Ascaris* (round worm) eggs had been found still living up to 107 days after inoculation. Toxacara can last more than 5 years .[127] Another study 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.[128]

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. A review of literature, suggests that the microbial hazard from faecal matter is modest.[129] 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.

The process of composting itself can also help to sanitise the final material through both pasteurisation (see Table B-15 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.[130] 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:[131]

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

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[126] I. Deportes, J.-L. Benoit-Guyod and D. Zmirou (1995). Hazard To Man And The Environment Posed By The Use Of Urban Waste Compost: A Review. Sci. Total Environ., 172: 197-222.

[127] I. Deportes, J.-L. Benoit-Guyod and D. Zmirou (1995). Hazard To Man And The Environment Posed By The Use Of Urban Waste Compost: A Review. Sci. Total Environ., 172: 197-222.

[128] MacKenzie D (1998). Waste Not in *New Scientist,* 29 Aug 1998 pp 26-30.

[129] I. Deportes, J.-L. Benoit-Guyod and D. Zmirou (1995). Hazard To Man And The Environment Posed By The Use Of Urban Waste Compost: A Review. *Sci. Total Environ*., 172: 197-222.

[130] I. Deportes, J.-L. Benoit-Guyod and D. Zmirou (1995). Hazard To Man And The Environment Posed By The Use Of Urban Waste Compost: A Review. *Sci. Total Environ*., 172: 197-222.

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

Table B-15: Temperature / Time Regimes Known to be Effective During Composting Against a Range of Pathogens of Importance to Humans

Disease	Organism	Lethal conditions
Non spore-forming bacteria		
Brucellosis	*Brucella abortus*	10 min: 60 °C
Cholera	*Vibrio cholerae*	15 min: 55 °C
Contagious abortion	*Vibrio fetus*	5 min: 56 °C
Diphtheria	*Corynebacterium diphtheriae*	10 min: 58 °C
Dysentery	*Shigella* spp	60 min: 55 °C
Food poisoning	*Salmonella* spp	20 min: 60 °C
Leptospirosis (Weil's disease)	*Leptospira* spp	10 min: 50 °C
Plague	*Yersinia pestis*	5 min: 55 °C
Staphylococcal infections	Staphylococci	30 min: 60 °C
Streptococcal infections	Streptococci	30 min: 55 °C
Tuberculosis	*Mycobacterium tuberculosis*	10 min: 60 °C
Typhoid fever	*Salmonella typhi*	20 min: 60 °C
Spore-forming bacteria		
Anthrax	*Bacillus anthracis*	10 min: 100 °C
Botulism	*Clostridium botulinum*	5 hr: 100 °C/5 min: 120 °C
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 of users against inherent pathogens. Provided all precautions are taken, the risk to humans or animals is believed to be minimal.

B.4.6 Plant Pathogens and 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 B-16, 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.[132] 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. Such features are increasingly being exploited also on a commercial level.

Table B-16: 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, C., E. Kron & S-E Svensson (1997). Compost Quality and Potential for Use AFR Report 154, Swedish Environmental Protection Agency

B.4.7 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);

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

- 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.

B.4.8 Heavy Metal Contamination in Soil

In the life-cycle context, the quality of compost is typically used to estimate emissions of heavy metals and persistent organic pollutants to water and land from compost. In practice, this is a highly flawed approach. The weightings for human toxicity impacts, for example, are not well understood, whilst the degree to which composts from source separated materials contribute in a negative way to human or environmental health is not clear. Metals may well become bound to humus so that they are effectively less likely to lead to problems.

Where composts are derived from source separated feedstocks, the level of contamination with potentially toxic elements is likely to be low.[133] Recent investigations have, indeed, sought to ensure that standards for quality composts are set to ensure no net accumulation of these contaminants in soil. Evidently, the issue is of greater significance – though it is no easier to analyse – where composts derived from mixed wastes are concerned.[134]

In most studies, heavy metals are considered such that the total amount fed into the compost equals the amount in matured compost. Because of the degradation of organic matter, the concentration of metals on a dry matter basis increases because of the mass loss in the process. The more mature composts will, other things being equal, show higher contaminant levels by this measure.

Where persistent organic pollutants (POPs) are concerned, different processes occur. Persistent organic pollutants appear to be affected by the compost process in different ways. Furthermore, different feedstocks appear more or less likely to carry high concentrations of POPs.[135] It is difficult to make generalisations concerning the concentrations of POPs in compost, still less to understand what their effects might be (with a view to subsequent monetisation of those impacts). Clearly, rates of application are also important.

No positive or negative impacts associated with heavy metals or POPs are attributed in this study. It would appear that any negative effects associated with compost applications would be associated with heavier PAHs and PCBs, and possibly, with PCDD/Fs. Positive effects may also arise in location-specific contexts due to the application of what are

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

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

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

pejoratively termed ‘heavy metals’, some of which (e.g. zinc and copper) are essential for plant nutrition, and depleted in some soils.

B.4.9 Non Monetised Benefits of Compost Utilisation

B.4.9.1 Physical and Biological Improvements to Soil

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[136] or physically in an attempt to separate it into functionally distinct types. It is generally considered that the organic matter light fraction, containing recognisable plant and animal debris, is the source of most of the readily mineralisable nutrients. The heavy fraction, more closely associated with the soil minerals, contains organic matter that is more highly humified (containing very large organic molecules with strong bonds) and stable over very long time periods. As discussed above, the composting process may produce organic matter which contributes to both these categories in the soil.

Although compost is high in organic matter it has yet to be proven that beneficial changes in the soil structure have been achieved following its application. Trials at IACR Rothamstead have shown an improved efficiency of mineral fertilisers where soil organic matter is high and this has certainly also been seen in other trials.[137] 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.

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

[137] D. S. Jenkins (1991) The Rothamstead Long-term Experiments: Are They Still of Use? *Agronomy Journal* 83.

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.

B.4.9.2 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.

B.4.9.3 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 750 kg N/ha). This precluded the use of lime to maintain pH in this trial.[138]

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.[139] 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.

B.4.9.4 Reduced Susceptibility to Soil Erosion

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

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

[138] L. Jackson, personal communication, based on series of reports, *Researching the Use of Compost in Agriculture 1997-2001*, HDRA Consultants.
[139] E. Favoino and M. Centemero (1995), Impiego di compost nella vivaistica: esperienze applicative svolte nel corso del 1994; in attachment to *Notiziario della Scuola Agraria del Parco di Monza*, May 1995.

alone were estimated at $2-$17 billion.[140] 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.[141]

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.

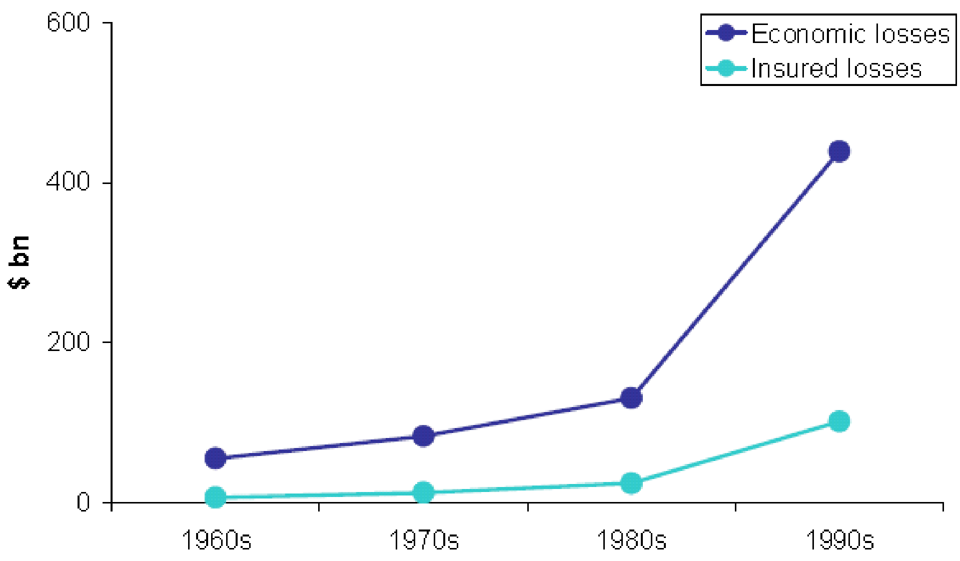
It is difficult (for obvious reasons) to estimate the incremental reduction in soil erosion associated with applications of compost. However, the benefits associated with reduced soil erosion are potentially significant. We have not quantified them here.

B.4.9.5 Reduced Risk of Flooding

Benefits from improved infiltration of water arise through reduced risk of flooding (and soil erosion – see above) and reduced requirement for irrigation water. Monetised benefits associated with the reduced requirement for irrigation are discussed in B.3.3.6. This section focuses on the reduced risk of flooding, for which no monetised benefit has been attributed.

A review of natural catastrophes published in 2004 suggests weather-related insurance damages have increased substantially over the past four decades. This is shown in Figure 11.

Figure 11: Cost of Weather Related Disasters Over Last 4 Decades (Adjusted to 2003 Prices)



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

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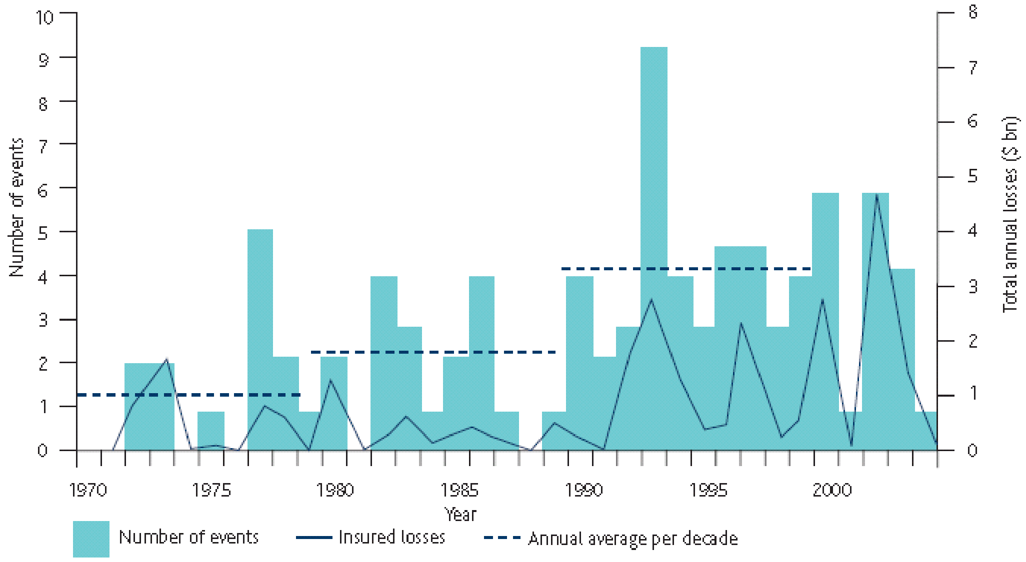
[140] National Research Council (1989) Problems in US Agriculture, in *Alternative Agriculture*, Washington DC: National Academy Press; M. Ribaudo (1989) Water Quality Benefits from the Conservation Reserve Programme, *Agricultural Economic Report No.606*, Washington DC: USDA Economic Research Service;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

[141] R. Evans (1996) *Soil Erosion and its Impact in England and Wales*, London: Friends of the EarthTrust.

In the UK, the DTI's Foresight Programme suggested that flood damages may increase twentyfold over 100 years.[142] It is quite clear that the insurance industry believes that changing risks associated with climatic events are driving insurance claims upwards, and flood damage is playing a significant part in this. Figure 12 depicts insured losses associated with flood events in Europe, Japan and the US, produced in a report for the Association of British Insurers. [143]

Figure 12: Number of Significant Events and Insured Losses, US, Europe & Japan (2004 Prices)



Source: Sigma Database, Swiss Re

Compost improves soil quality and increases the capacity to hold moisture. Quantification of any benefit here is however 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).

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

B.4.9.6 Bioremediation of Soil Using Compost

Compost and the composting process can be used successfully in the bioremediation of contaminated soils. In-situ remediation is commonly used whereby compost is used essentially as an inoculate to the contaminated soil, providing the microorganisms which

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[142] See Norwich Union Risk Services (2004) The Cost of Floods Could Escalate, http://www.nu-riskservices.co.uk/news/articles/cms/1084110533626421411811 4_1.htm, 23/04/2004.

[143] Climate Risk Management and Metroeconomica (2005) *Financial Risks of Climate Change*, Report to the ABI, June 2005.

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.

B.4.9.7 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.

B.4.9.8 Non Monetised Benefits Associated with Reduced Peat Use

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

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

The last of these has already been dealt with B.3.4.2. This section discusses the other environmental costs of peat extraction which have not been monetised in the present study.

Loss of Biodiversity

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

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 translates values for the Somerset Levels into a value of £7,245 per hectare.[144] Another study estimated a preservation value of £68.4 million, or £4.1 million per annum using a 6% discount rate.[145]

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

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

Loss of Paleological and Archeological 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.

C Anaerobic Digestion

Our study considers three options for use of the biogas:

1. The biogas is used to generate energy as either electricity, heat or a combination of the two. In this case the biogas is usually used on site;
2. The second, becoming more common, is the use of biogas to power vehicles. In this case, the biogas is usually cleaned prior to its utilisation as compressed natural gas.
3. Less commonly, the biogas can be further cleaned and injected into the natural gas grid.

C.1 Factors Affecting Plant Performance

C.1.1 Biogas Generation

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

For biological materials, the carbon content and the volatiles solids content of the biowaste are related by the following equation:[146]

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

Reactor performance is often related to the rate of volatile solids destruction. For digesters, the figure usually lies between 50-70%. This figure can vary in accordance with the type of reactor, the retention time and the nature of the biodegradable wastes entering the reactor. Performance is often measured in terms of methane production per unit of volatile solids input to the digester. For example, Vandevievre et al report variation from 90 Nm^3 CH_4 per tonne fresh garden waste to 150 Nm^3 CH_4 per tonne fresh food waste (210-300 Nm^3 CH_4 per tonne VS, equivalent to a 50%-70% reduction in VS). Knowledge of one or other of the carbon content or the volatile solids content allows an estimation of the methane generated. Since the methane content of biogas is usually 50%-65% of the total biogas and since the remainder is principally carbon dioxide, basic estimates of CO_2 and methane generation in raw biogas can be made.

Biogas generation varies considerably across different facilities. One study from Canada (Chavez-Vasquez and Bagley, 2002) sought to relate the production of biogas from different processes to the different fractions of the waste stream. The results are shown in Table C-1. From this, it seems quite clear that performance varies significantly across digestion facilities depending upon their design and operational characteristics.

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

Table C-1: Digester Performance from Different Reactor Types

	m³ biogas per tonne of volatile solids					
	One-stage High solids			One-stage Low Solids	One-stage High Solids (Batch)	Two-stage High Solids (with Steam Disruption)
	Valorga	Kompogas	Dranco	BTA	Biocel	SUBBOR
Kitchen waste	634	460	671	695	498	880
Garden waste	211	152	426	230	166	293
Paper and Cardboard	353	256	485	385	277	490

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

The ORWARE model also tries to predict the biogas generation from the composition of input wastes. It is based upon a plant in Uppsala described by Dalemo.[147] This was a one-stage wet CSTR plant. In ORWARE, the amount of biogas from anaerobic digestion depends of the incoming materials composition of organic carbon. The model assumes degradation of organic substances according to first order kinetics:

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

Where:

D = Degradation ratio

D_0 = Maximum degradation ratio

k = first order rate constant

HRT = Hydraulic retention time

Different organic substances are decomposed at different rates depending upon the constituent organic compounds. The parameters used are shown in Table C-2.

Table C-2: Performance Parameters for the Anaerobic Digestion Process

Organic substances	D_0	K (days^{-1})	% CH_4
C_{-chsd} (lignin)	0	0.001	50
C_{-chmd} (cellulose)	1-1.77* C-chsd	0.18	50
C_{-chfd} (starch. sugars)	1.0	0.23	50
C_{-fat}	0.95	0.13	69
$C_{-protein}$	0.8	0.13	78

Sources: M. Dalemo (1997) The ORWARE Simulation Model - Anaerobic Digestion and Sewage Plant Sub-models. Licentiate thesis. Swedish University of Agricultural Sciences, SLU, Uppsala. M. Dalemo. (1999). Environmental Systems Analysis of Organic Waste

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

Management. The ORWARE Model and the Sewage Plant and Anaerobic Digestion Submodels. Ph D Thesis. Swedish University of Agricultural Sciences, Uppsala.

It should be noted that the work upon which the study was based – outlined in more detail by Dalemo – made reference to the validity of the data. It noted that the data was applicable to CSTR systems and that the rates of degradation were applicable to the mesophilic range. It is not clear how these data were adapted (if at all) by Baky and Eriksson. Dalemo also includes some modelled figures for digestion of source separated household waste and other materials using data on organic composition from Sonesson and Jonsson.[148] These estimates of biogas production are shown in Table C-3.

Table C-3: Modelled Performance of Reactors

		Source separated household waste	**Source separated waste from restaurants**	**Manure[1]**	**Slaughter-house Waste**
Content					
Protein	% DM	13	13	8.8	50
Fat	% DM	18	24	0.7	37
Fast degrading carbohydrates	% DM	22	19	0	2.7
Medium degrading carbohydrates	% DM	23	20	57	0
Slow degrading carbohydrates	% DM	4.4	4	8	0
TS	%	35	30	1.6	30
Gas Production					
HRT=10	l CH_4/kg VS	210	190	155	160
HRT=15	l CH_4/kg VS	330	360	210	480
HRT=20	l CH_4/kg VS	360	395	230	520
HRT=30	l CH_4/kg VS	390	430	250	570
Gas Composition	% CH_4	61	63	55	72

Notes:

1. circa 50% manure from pigs and 50% from cattle

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

[148] U. Sonesson and H. Jonsson (1996) Urban Biodegradable Waste Amount and Composition – Case Study Uppsala. Dept. of Agricultural Engineering, Swedish University of Agricultural Sciences, Uppsala.

Even though the ORWARE model is capable of relating biogas generation to input materials, Baky and Eriksson essentially restricted the generation of biogas to 125 Nm^3 per tonne of waste input.[149] They assumed 50% thermophilic and 50% mesophilic operation. This is a low rate of biogas generation.

Finnvenden et al[150] based their inventory data on a study by Nilsson.[151] The inventory data was collected at the digestion plant in Kristianstad, Sweden. The plant was a wet one-stage mesophilic anaerobic digestion process, and was receiving organic waste mainly from households, food industry, restaurants and agriculture. In their study, the original inventory data for anaerobic digestion was supplemented with data on metals according to the food waste composition. The metals were assumed to be retained within the digestion residue.

A weight loss of 40% of the total solids (TS) in food waste (assumed 30% dry matter) was assumed due to degradation through release of CO_2 and CH_4 during the digestion process.

Organic wastes are inherently very variable in composition. For example, whilst canteen wastes have a dry matter content of around 30%, fruit and vegetable wastes (e.g. from markets) may be as much as 90% moisture.[152] The solubility of the total solids content is also important in defining the ability of the feedstock to degrade in a digester. Both these factors will influence the amount of biogas (and CH_4) produced from a given feedstock.

Approach Taken in This Study

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

- The ORWARE approach, assuming a hydraulic retention time of 18 days;
- An approach based on the work of Chavez-Vasquez and Bagley discussed above; and
- An approach based on VS content and using assumptions concerning the rate of VS destruction and the rate of conversion of VS into methane from Cecchi et al[152].

We then compared these results with those from suppliers which we gathered in the context of work carried out in Northern Ireland (see Table C-4).

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

For food wastes, and following the approach taken by Cecchi et al, we therefore assume a VS loss of 60%, and that the biogas produced contains 60% CH_4. This leads to a biogas production of 500 m^3 per tonne of VS. This biogas production is representative of a substrate containing 30% dry matter – i.e. canteen wastes, or food wastes mixed with a small amount of garden waste.

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

[150] Goran Finnvenden, Jessica Johansson, Per Lind and Asa Moberg (2000) *Life Cycle Assessments of Energy from Solid Waste*, Forskningsgruppen for Miljostrategiska Studier, FMS 137, August 2000.

[151] B. Nilsson (1997). Kompostering eller rötning? En jämförande studie med LCA-metodik. Master Degree Thesis. Chalmers Tekniska Högskola, Göteborg.

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

Table C-4: Biogas Production in Plants

Plant	1	2		3		4	5
Feedstock	**Kitchen and Garden**	**Kitchen and Garden**		**Kitchen and Garden**		**Kitchen**	**Kitchen and Garden**
Retention Time (days)	21	16	21	14	16	25	25
Gas production per kilogram wet waste (m^3 biogas/tonne input waste)	120	110	150	122	138	140	170
Methane content of biogas (%)	55%	56%	62%	60%	65%	61%	59%
Methane production per tonne wet waste (m^3 methane / tonne input waste)	66	75		76	86	85	100

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

Cecchi et al looked at the relative performance of an AD process using two different feedstocks, with comparisons made in terms of the loss of VS:[153]

- A source separated organic stream, which from the high moisture content (over 80%) is assumed to be food waste, leading to a VS loss of 70%; and
- A municipal waste stream (MSW), which led to a VS mass loss of 37%.

Cecchi's study did not provide any composition data for the MSW fraction. However it is likely that much of the biogenic carbon will be contained within the food waste and paper fractions, with the remainder found in textiles made from natural fibres and garden wastes. The last of these contains appreciable quantities of lignin which is also found in some types of paper (principally newspaper).

Lignin does not degrade anaerobically; as such it has been suggested that the lignin content of the substrate might be used as an indicator of the degradability of a waste feedstock within AD.[154] Other authors suggest that biodegradability is strongly dependent on the structure of the ligno-cellulosic compounds contained within the material, and that the lignin content is therefore not the sole indicator of feedstock behaviour.

Data from Chavez-Vasquez and Baguely (see Table C-4) suggest that the biogas production from garden wastes is only a third of that produced from the degradation of food wastes, although data is highly variable between the different AD processes. They further suggest feedstocks of paper / card will produce approximately half of the biogas generated by kitchen wastes.

Typically, more than two thirds of the organic fraction of MSW is food and paper / card. Since the organic fraction of MSW contains more of the materials that are likely to degrade anaerobically, we would therefore expect a reduction in the VS loss resulting from the AD of garden wastes in comparison to that seen with a MSW feedstock.

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

[154] Sanders W, Veeken A, Zeeman G and van Lier J (2003) Analysis and Optimisation of the Anaerobic Digestion of the Organic Fraction of Municipal Solid Waste, in J. Mata-Alvarez (ed) (2003) *Biomethanization of the Organic Fraction of Municipal Solid Wastes*, London: IWA Publishing, pp.141-179

Cecchi et al also suggest that the CH_4 content of the biogas produced from a MSW feedstock is lower than that produced by the source separated organic stream – whilst the latter generates a biogas containing 60% CH_4 this proportion is reduced to 50% where MSW is used as the feedstock.

Taking into account the above data, we have made the following assumptions in modelling the behaviour of a source separated garden waste stream within an AD facility:

- A VS loss of 20%; and
- A biogas CH_4 content of 50%.

These assumptions result in a biogas production of 200 m^3 per tonne of VS, which is slightly lower than the mean of digester performance figures seen in Table C-4 where garden waste is the substrate.

C.2 Direct Emissions to Air from the Process

The ultimate emissions to atmosphere, and indeed, the emissions associated with the compensatory system, are different depending upon the utilisation of the biogas. However emissions associated with AD are also likely to occur at the following points in the process:

1. during the digestion phase;
2. during the stabilisation process used to treat the solid residue.

Emissions associated with the utilisation of the biogas (including those associated with the combustion of the biogas) are discussed in Section C.4. The following section discusses those emissions that occur elsewhere in the process, at the points outlined above.

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

C.2.1 Climate Change Impacts

Emissions are less likely during the digestion phase as the process is fully enclosed. Fugitive emissions (such as those that may occur from the shredding and sorting of the waste) are also likely to be minimal in well-run facilities.[155] Data regarding these emissions is however relatively limited.

The Reference Document on the Best Available Techniques for the Waste Treatment Industries presented data on emissions from the digestion process, collected from a number of sources. Emissions relating to climate change are presented in Table C-5. This suggests fugitive emissions of CH_4 to be less than 1% of the total CH_4 content of the biogas. The data in the table includes emissions associated with energy generation (discussed in Section C.4).

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[155] European Commission (2006) *Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for the Waste Treatment Industries*, August 2006

Table C-5: Emissions Data from Digestion Process

Pollutant	Example emissions data	
	Value	Unit
Methane	0-411	g / tonne
CO_2	181,000 - 520,000	g / tonne
N_2O	Assumed minimal	
Note CO_2 emissions result from the combustion of the biogas (discussed in Section C.4)		

Source: European Commission (2006) Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for the Waste Treatment Industries, August 2006

A more recent German study obtained data from five Bavarian agricultural biogas facilities for one year, with the aim of establishing greenhouse balance of these plants.[156] The study commented that fugitive emissions of CH_4 "eluded concrete measurement" but estimated diffuse CH_4 emissions to be 1%, equivalent to around 600 g of CH_4 per tonne of waste. Additional CH_4 emissions were found to occur during the energy generation phase, which, for the plant under investigation, occurred on-site using a gas engine.

The same study found that the fermentation residue emitted CH_4 whilst being stored, although the authors noted that these emissions were minimised by covering the storage container and capturing the gas (this occurred in only one plant of the five studied). Emissions ranged from 30 – 130 g CO_2 equivalent per kWh of electricity. The study estimated an average of 70 g CO_2 equivalent per kWh of electricity, equivalent to approximately 800 g of CH_4 per tonne of waste to the facility. It is not clear whether post-digestion treatment emissions are included within the emissions data presented in Table C-5.

The present analysis assumes that the residue of the digestion process is treated using a stabilisation process, and this will also result in reduced emissions occurring during the post digestion phase. Section B.1.2.2 suggests CH_4 emissions resulting from the stabilisation process will be around 600 g CH_4 per tonne of input to the process assuming a well-managed composting plant. Emissions resulting from the stabilisation of digestion residue are likely to be less than this, given that the input material will have already passed through the digester and will contain substantially less carbon.

Approach Taken in This Study

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

Our model assumes fugitive emissions of 900 g of CH_4 (per tonne of waste entering the facility) are emitted from the digester and from the post-digestion phase. Further CH_4 emissions will result from the energy generation phase and these are discussed in

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

Section C.4. Table C-6 summarises the key assumptions used to model the digestion phase and the associated climate change impacts.

Table C-6: Assumptions Relating to AD Process and Generation of Biogas

Parameter	Assumption
Dry matter content of food waste	30%
Organic matter content of VS	93%
Carbon content of VS	45%
VS content of organic matter	45%
VS loss during digestion	70%
Methane content of biogas	60%
Fugitive CH_4 emissions from digester (% carbon converted to CH_4)[1]	1.5%
CO_2 emissions from the process (kg CO_2 per tonne of waste input)	276

Notes

1. Equivalent to 900 g of CH_4 per tonne of waste to process. Further CH_4 emissions will occur during combustion or upgrading of the biogas (discussed in Section C.4)

C.2.2 Air Quality Impacts

Table C-7: Emissions Data from Anaerobic Plants

Pollutant	Example emissions data	
	Value	Unit
CO	72.3	g / tonne
NOx	10-72.3	g / tonne
NH_3	Fugitive	
SOx	2.5-30	g / tonne
H_2S	284-289	mg/Nm^3
TOC (VOC)	0.0023	g / tonne
Odour	626	GE/Nm^3
Dioxins/furans	10-8	g / tonne
Total chlorine	1.5	µg/Nm^3
HCl	0.011	g / tonne
HF	0.0021	g / tonne
Cd	9.4E-07	g / tonne
Cr	1.1E-07	g / tonne
Hg	6.9E-07	g / tonne
Pb	8.5E-07	g / tonne
Zn	1.3E-07	g / tonne

Source: European Commission (2006) Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for the Waste Treatment Industries, August 2006

Data relating to air quality impacts associated with the digestion phase is very limited. It is generally accepted that trace elements of gases such as hydrogen sulphide (H_2S) will be present in the gas generated by anaerobic digesters. Fugitive emissions of this gas may also occur, although these are likely to be minimal in well managed facilities.

H_2S is usually removed from the biogas to reduce emissions from the overall system (i.e., including those that occur during the subsequent energy generation phase). This can be achieved either using activated carbon filtration or by scrubbing with iron salts.

Table C-7 presents example emissions data for anaerobic plants as outlined in the previously cited Reference Document for the Waste Treatment Industries. This data includes emissions from both the energy generation phase (likely to involve the combustion of the biogas in a gas engine) as well as those originating from the digester.

Of the data presented in the table, H_2S and VOC emissions may result from the digestion phase, whilst the NOx, SOx and CO emissions are more likely to be associated with the combustion of the biogas. The data in the table relates to the digestion of MSW as part of an MBT process. VOC emissions from the anaerobic digestion of source separated food wastes are typically less than those that result from a MSW feedstock.

Approach Taken in This Study

Although fugitive emissions of H_2S may occur in digestion plants, no damage cost is associated with this pollutant. Our model therefore attributes no external cost to air quality impacts occurring during the digestion phase. We further assume that the VOC emissions associated with this part of the process will be negligible in a well managed facility, due to the enclosed nature of the process. Emissions occurring during the energy generation phase result in a more significant influence on the external costs resulting from impacts to air quality, and these are discussed in Section C.4.

C.3 Energy Used at Facilities

Unlike composting plant, AD facilities can potentially utilise some of the energy generated within the process to meet their requirements, although the literature suggest that some plant supplement this with electricity taken from the grid. AD facilities typically use both electricity and heat, discussed in the two sections that follow.

C.3.1 Electricity

Data provided from the previously cited study of five Bavarian agricultural biogas plant suggested that these facilities utilised on average 5.9% of the electricity generated for their own internal purposes (although plant surveyed did not always use their own electricity).[157] This suggests an electricity requirement of 17 kWh per tonne of waste to the process.[158]

Another German study presented results of an on-going investigation into the energy input associated with feeding systems for AD facilities and noted that:[159]

> *…the internal power consumption of biogas plant varied between 4% and 13% of the electricity generated. Almost 25% of plants consume 8-9% of the electricity they generate.*

Elsewhere, data from the UK biogas technology supplier Greenfinch suggests that between 3 and 28 kWh of electricity per tonne of input was required by the process,

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

[158] This assumes a gross electrical generation efficiency of 37%, as discussed in Section C.4.1.

[159] J. Sedlmeier (2008) Modern Feeding Systems – A Comparison of Energy and Time Input, *17th Annual Convention of Fachverband Biogas e.V,* 15th-17th January 2008, Nuremberg

depending on the feedstock (although a lower electricity generation efficiency was indicated for this process).[160]

There will be an additional electricity requirement associated with the gas upgrading process where the intention is to use the biogas as a vehicle fuel or to inject it into the gas grid. These demands may vary depending on which upgrading process is used. One study suggests that the upgrading steps require an input of electricity amounting to around 6% of the energy produced.[161] More recent data from both Germany and Sweden suggests that this is typically in the order of 0.2 kWh per Nm^3 of biogas.[162]

C.3.2 Heat

Commenting on the heat requirements of the AD process with regard to the treatment of MSW, the previously cited Reference Document for the Waste Treatment Industries suggested that up to one third of the biogas produced is needed to heat the digester itself.[163]

The significant heat requirement is confirmed by data provided from plant in Germany and the UK. Data from Bavaria suggests that a maximum two thirds of the heat produced by agricultural biogas plants can be used in some way under practical operating conditions, suggesting a heat requirement of 125 kWh per tonne of input.[164] Mass balance information provided by the UK operator Greenfinch suggests that up to 50% of the heat may be required (equivalent to 216 kWh of heat per tonne of input) although 130 kWh is more typical.[165]

Approach Taken in This Study

The current study assumes that the AD process utilises 30 kWh of electricity and 118 kWh of heat per tonne of input to the process, equivalent to 10% of electricity generated, and 33% of the heat generation. The additional electricity requirement for upgrading is assumed to be 28 kWh of electricity (equivalent to 0.2 kWh per Nm^3 of biogas).

These energy requirements are assumed to be supplied by the AD process itself – i.e. a smaller CHP unit is assumed to fuel the vehicle fuel and gas to grid applications. The energy content of the biogas that is assumed to be available for the upgrading process is therefore reduced accordingly.

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

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

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

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

[163] European Commission (2006) *Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for the Waste Treatment Industries,* August 2006

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

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

C.4 Energy Generation

As was indicated at the outset of this section, the current study considers the following uses for the biogas produced from the AD process:

1. to generate electricity or heat on site using a gas engine;
2. upgraded biogas used to power vehicles;
3. upgraded biogas injected into the natural gas grid.

Each will result in emissions from the utilisation process itself, as well as offsetting emissions associated with the energy generation that would have otherwise taken place. The ultimate emissions to atmosphere, and indeed, the emissions associated with the compensatory system, will be different depending upon the utilisation route. Avoided emissions are also dependent upon the energy mix of the country, as was discussed in Section A.4.

C.4.1 On-site Combustion of Biogas

Data from Greenfinch suggests a gross electrical generation efficiency of 30% together with a heat generation efficiency of more than 50%.[166] More recent data from the same technology producer suggests higher electrical generation efficiencies along with lower heat generation efficiencies. This more in line with data from Germany, which suggests a gross electrical generation efficiency of 40% together with a gross heat generation efficiency of 45%.[167]

Our study assumes a gross electrical generation efficiency of 37% and a gross heat generation efficiency of 45%. A proportion of the energy generated is assumed to be consumed by the process, as was discussed in Section C.3. 60% of the *net* heat generated is assumed to be utilised, as was discussed in Section A.4.5.

C.4.1.1 Climate Change Impacts

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

As was previously indicated in Section C.2.1, additional CH_4 emissions result from the non-combusted gas (known as the "slip") from gas engine. Data from five Bavarian agricultural biogas plant suggests this results in emissions of 21-37 g CO_2 equivalent per kWh electricity.[168] N_2O emissions from the combustion of biogas are assumed to be negligible.

As previously indicated in this section, biogas combusted in a gas engine is assumed to result in the net generation of 268 kWh of electrical energy, and 102 kWh of heat (taking into account the utilisation factor). The electricity generation results in offset emissions of between 5 and 228 kg CO_2 equivalent depending on the energy mix of the country. A further offset of 19-32 kg CO_2 equivalent is possible where the heat is also exported.

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

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

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

C.4.1.2 Air Quality Impacts

Where biogas is used to generate energy using a gas engine, emissions are associated with the combustion process. The most significant air quality impacts resulting from the combustion of biogas relate to emissions of SOx and NOx.

SO_x emissions are generally a matter for process management. To the extent that they stem from the H_2S in raw gas, the emissions are related to the use of, for example, precipitation salts which seek to precipitate out the sulphur emitted from the degradation of proteins as iron sulphide.

NOx emissions result from high temperature reactions occurring between atmospheric nitrogen, and trace elements such as NH_3 and H_2S contained within the biogas.[169] These emissions can be significantly reduced using Selective Catalytic Reduction (SCR) techniques. Use of this technique was cited as the Best Available Technique (BAT) in the Reference Document for the Waste Treatment Industries for MBT systems using AD. However, although this equipment is widely used to reduce NOx emissions from waste incineration facilities across Europe, use of SCR within AD plant is still relatively limited.[170] The air quality limits imposed upon AD facilities are typically less onerous than those for waste incinerators; thus there is a lack of incentive to incur the further expense that the installation of this additional equipment requires.

Table C-8 presents data from the aforementioned Reference Document for emissions resulting from the combustion of biogas in a gas engine. The Document gives emissions data expressed in terms of the concentration of pollutants within exhaust gas. Emissions per tonne of waste to the process are calculated in Table C-8 using data supplied from the GRL MBT facility at Eastern Creek (produced as part of their emissions testing regime).[171]

The data in Table C-8 suggests that the emissions examples previously presented elsewhere within the Reference Document considerably underestimated the total air quality impacts resulting from the whole AD process (previously shown within this report in Table C-5). In addition, the Document indicates that the example emissions data was calculated with reference to a exhaust gas volume of 11,000 m^3 per tonne of waste to the facility – considerably greater than that suggested by the GRL test report. The greater volume of exhaust gas would lead to even larger emissions (on a per tonne basis) than those currently presented in Table C-8 which were calculated using the GRL exhaust gas data.

The GRL test report indicates NOx emissions of 350 mg/m^3, and SOx emissions of 10 mg/m^3 for the 1,205 kWh gas engine under inspection. Although biogas at the facility is scrubbed prior to the energy generation process resulting in reduced SOx emissions, the plant does not employ SCR to reduce emissions of NOx.

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[169] V. Aschmann, R Kissel and A Gronauer (2008) Efficiency and Environmental Compatibility of Biogas Fired Cogeneration Plants in Practical Service, *17th Annual Convention of Fachverband Biogas e.V,* 15th-17th January 2008, Nuremberg

[170] It is understood, however, that authorities in The Netherlands wanted to impose this as standard for all AD facilities.

[171] VOC emissions occurring during the digestion phase may be higher in AD plant treating MSW in comparison to those seen in facilities treating source separated material. However, emissions occurring at the biogas combustion stage are likely to be similar for facilities treating either feedstock, as the biogas must be relatively clean in order to ensure the smooth operation of the gas engine.

Table C-8: Emissions Values for Biogas Combustion Using Gas Engine

	Exhaust gas emissions, biogas combustion (mg/m^3)	Equivalent emissions (g / tonne of waste)
CO	100 - 650[1]	68 - 441
Dust	< 50	< 34
NOx	100 - 500[2]	68 - 339
H_2S	< 5	< 3
HCl	< 30	< 20
HF	< 5	< 3
Hydrocarbons	< 150	< 102
SO_2	50 – 500	34 - 339

Notes

Exhaust gas emissions data assumes 5% oxygen content.

Equivalent emissions are calculated assuming 3,960 m^3 exhaust gas is produced per hour from each of two gas engines, using data provided by GRL from their Eastern Creek MBT facility in Australia (that facility treats 70,000 tonnes of waste per year in the AD part of the plant).

1. When using spark engines with low heat capacity (e.g. < 3 MW_{th}) the value of 650 may be difficult to achieve. In those cases 1,000 mg/m^3 is seen as more achievable.
2. When using pilot injection engines with a low firing capacity (e.g. < 3 MW_{th}) the achieved values are 1,000 mg/m^3. The lower end of the range can only be achieved with abatement techniques (SCR).

Sources: European Commission (2006) Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for the Waste Treatment Industries, August 2006; EML Air PTY (2008) Test Report Prepared for Eastern Creek Operations, report for Global Renewables Ltd, April 2008; AEA (2007) Mechanical Biological Treatment: Case Study 2: Eastern Creek UR 3R Sydney, Report to IEA Bioenergy Task 36, April 2007

Data from German biogas plant confirms that the exhaust gas produced by smaller facilities often contains much higher concentrations of NOx than that seen at the Eastern Creek facility.[172] Analysis of the exhaust emissions produced by engines ranging from 110 to 526 kWh_e in size indicated no case where the NOx emission concentration was less than 500 mg/m^3. At the upper end of the range, emissions concentrations of more than 2,000 mg/m^3 were seen. Performance was generally improved after maintenance, resulting in emissions reductions of 50% in some cases.

Against this, the electricity generation from one tonne of food waste to the AD facility results in offset emissions of between 19 and 705 g of NOx, depending on the energy mix of the country. A further offset of 26-50 g NOx is possible where the heat is also exported. Emissions of SOx from energy generation are also significant in those countries reliant on oil and coal as part of their energy mix, resulting in further avoided emissions.

[172] V. Aschmann, R Kissel and A Gronauer (2008) Efficiency and Environmental Compatibility of Biogas Fired Cogeneration Plants in Practical Service, *17th Annual Convention of Fachverband Biogas e.V,* 15th-17th January 2008, Nuremberg

Approach Taken in This Study

The current study assumes biogenic CO_2 emissions of 276 kg CO_2 per tonne of waste to facility result from the combustion of the biogas in the gas engine. We further assume emissions of 30 g CO_2 equivalent per kWh of electricity from the non-combusted biogas resulting in 8 kg CO_2 equivalent emissions from one tonne of waste to the process.

Other emissions from the combustion of biogas are based on emissions data from the GRL facility, supplemented by data provided in the Reference Document for those pollutants not included within the test report. These assumptions are outlined in Table C-9.

Table C-9: Emissions from the Combustion of Biogas

	Emissions from biogas combustion (g / tonne of waste)
CO	200
Dust	20
NOx	250
Hydrocarbons	40
SO_2	20

C.4.2 Use of Upgraded Biogas as a Vehicle Fuel

The utilisation of biogas as vehicle fuel uses the same engine and vehicle configuration as natural gas. There are reportedly more than 1 million natural gas vehicles in use across the world, which demonstrates that there is a receptive market to the use of biogas as vehicle fuel.[173]

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

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

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

C.4.2.1 Biogas Upgrading Processes

Gas quality demands are strict so as to provide a consistent high calorific gas containing low levels of contaminants and corrosive gases. The raw biogas produced in AD plants contains CH_4 and CO_2, smaller amounts of H_2S and NH_3, and trace amounts of H_2, N_2, CO, and O_2. Across different countries the minimum CH_4 content specification is between 95% and 97%, the permissible remainder being mostly CO_2. Typically also, the vapour

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

content must be lower than 15 mg/Nm^3, the H_2S content should not exceed 100 mg/Nm^3 and the particle size is limited at 40 microns. The typical sequence for gas preparation is:[174]

- A two step biogas desulphurisation process involving firstly a coarse desulphurisation method such as sulfide precipitation, followed by a fine desulphurisation step typically using activated charcoal filters;
- Gas drying;
- Gas compression;
- Removal of the CO_2 from the biogas (sometimes called CO_2 sequestration). This is most commonly achieved by scrubbing the gas with water under pressure, although other methods such as Pressure Swing Adsorption (PSA) are also used.

The CO_2 removal step results in the loss of some CH_4 from the biogas. These losses are typically in the order of 1% for the water scrubbing methods, although they can be as much as 3% if PSA is used. However some technology providers claim they can reduce this amount to close to zero.[175] The clean up process is also associated with an additional energy, as was previously discussed in Section B.2.1 .

C.4.2.2 Climate Change Impacts

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

Natural gas has a lower carbon content than diesel, which results in a reduction in greenhouse gas emissions where these are calculated on the basis of the amount of energy consumed. Gas also has a high octane number, enabling a high compression ratio to be used, further reducing emissions.[176] However gas-fuelled vehicles emit more CH_4 than diesel vehicles. In addition, differences in fuel consumption between the two types of vehicles may reduce the benefits seen when emissions are calculated per km of travel. Data from France suggests the fuel consumption for the biogas buses is 65 m^3 per 100 km, whilst diesel buses use 50 litres per 100 km.[177] This gives fuel consumption for biogas buses of 23.4 MJ per km, whilst that of diesel vehicles is 17.9 MJ per km.

A study in Finland by VTT compared the emissions performance between a number of diesel and CNG buses, as part of a comprehensive national programme investigating bus emissions. Their analysis considered vehicles in prime condition manufactured during 2002-4, representative of Euro III technology. Data from that study with regard to the greenhouse gas emissions is presented in Table C-10.

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

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

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

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

Table C-10: Emissions Data from Diesel and Gas Buses

		Emissions, g / km	
		CO_2	CH_4
Diesel	Euro III	1,150	0.01
	Euro III + OC	1,200	0.01
	Euro III + CRT	1,230	0.05
Gas	Euro III + LB CNG	1,230	0.60
	EEV LB CNG + OC	1,420	1.90
	EEV LM CNG + TW/OC	1,300	0.30
	EEV SM CNG + TW	1,050	1.20

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

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

However, an earlier report detailing tailpipe emissions from Swedish buses suggests much lower emissions from gas buses, giving values of 524 g CO_2 per km for a natural gas bus and only 223 g / km for a biogas bus.[178] It is not clear whether the last figure includes the biogenic CO_2 emissions; if this is not the case, the CH_4 emission is far larger than anything seen within the VTT test data. The same study suggested emissions from a diesel bus of 1,053 g / km, which is more in line with the VTT dataset.

Other data produced by car manufacturers suggests that the use of dual fuelled cars operating with a mixture of natural gas and diesel results in emissions reductions of 10-15% in comparison to comparable petrol fuelled vehicles.[179] Those vehicles fuelled solely by gas are anticipated to achieve greater emissions reductions. More recently published data associated with a planned trial for dual fuelled buses in the UK indicates that carbon dioxide emissions reductions of 14% are expected as a result of the shift to the dual fuel vehicles.[180]

C.4.2.3 Air Quality Impacts

Gas fuelled vehicles typically emit less NOx when compared to petrol or diesel fuelled vehicles. Data from car manufacturers suggests that SOx and particulates are virtually eliminated, and that unburnt hydrocarbons (such as CH_4 contribute less to tropospheric ozone formation than do the volatile organic compounds present in petrol exhaust emissions.[179]

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[178] C. Plombin (2003) Biogas as Vehicle Fuel: A European Overview: Trendsetter Report No 2003:3, Stockholm
[179] See http://www.whatgreencar.com/cng.php
[180] See http://www.letsrecycle.com/do/ecco.py/view_item?listid=37&listcatid=217&listitemid=53457&section=waste_management

Emissions data is available from gas buses operating in several European countries. Data from the previously cited Finnish study produced by VTT is shown in Table C-11.

Table C-11: VTT Emissions Data for Diesel and Gas Fuelled Buses

		Emissions g / km		
		NMVOC	**NOx**	**PM10**
Diesel	Euro III	0.30	7.5	0.15
	Euro III + OC	0.10	8.2	0.12
	Euro III + CRT	negligible	8.6	0.02
Gas	Euro III LB CNG	negligible	9.2	<0.01
	EEV LB CNG + OC	0.05	4.5	0.02
	EEV LM CNG + TW/OC	negligible	2.0	<0.01
	EEV SM CNG + TW	negligible	2.1	0.01

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

Table C-12 presents data taken from buses operating in the city of Gothenburg, in Sweden which also suggests a reduction in NOx and particulates might be expected where buses are fuelled with either biogas or natural gas. The Swedish dataset gives no indication of the type of engine used for any of the vehicles although the date of the document suggests these are likely to be Euro III or earlier.

Table C-12: Bus Emissions Data from Sweden

	Emissions g / km		
	NMVOC	**NOx**	**Particles**
Diesel	0.40	9.73	0.100
Natural gas	0.60	1.10	0.022
Biogas – Gothenburg buses	0.35	5.44	0.015

Source: C. Plombin (2003) Biogas as Vehicle Fuel: A European Overview: Trendsetter Report No 2003:3, Stockholm

The data from the Gothenburg buses indicates that NOx emissions from upgraded biogas are higher than those for natural gas fuelled buses. However more recent data presented in Table C-13 taken from similarly fuelled buses operating in Lille suggests lower emissions of NOx for buses fuelled with upgraded biogas but an increase in particulates. Table C-13 also confirms the improvement in emissions seen as a result of the change from Euro III to Euro IV standard engine for the diesel-fuelled vehicles. The Lille study did not provide any information as to the type of engine used in the case of the biogas fuelled bus.

Table C-13: Bus Emissions Data from France

	Emissions g / km		
	NMVOC	NOx	Particles
Diesel (Euro IV)	0.46	3.50	0.020
Biogas – Lille buses	0.37	1.59	0.042

Source: Bio-NETT (2008) Bio-methane in Lille: A Case Study

Approach Taken in This Study

Our study assumes that the upgraded biogas is used to fuel heavy goods vehicles such as a bus or a waste collection vehicle, as is the case in Sweden, France and the UK. Comparisons are made on the basis of emissions reductions per km of travel, assuming the same type of engine is used for both the diesel and gas fuelled vehicles. We assume that the use of biogas results in emissions reductions of:

- 15% for the greenhouse gases (in terms of CO_2 equivalent emissions);
- 75% for the NMHC;
- 46% for NOx; and
- 90% for particulates.

These figures are based on the anticipated emissions reductions for the UK biogas buses with respect to greenhouse gas emissions, and the measured emissions reductions seen for biogas-fuelled buses in Sweden and Finland for the other air pollutants. Use of biogas is also considered to offset emissions associated with the production of the diesel, discussed in Section A.4.4.3.

We also assume that 2% of the CH_4 in the biogas is emitted during the upgrading process.

C.4.3 Injection of Upgraded Biogas into the Gas Grid

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

Gas distribution grids in many European countries are divided into L-gas grids and H-gas grids. For some such as France, this denote grids accepting gas of a different calorific values whilst in others such as Germany it reflects gases of different CV and Wobbe index values, where the Wobbe Index is the performance standard that defines the quality of the gas with respect to its use in appliances.

In Germany, biogas injected into an L-gas grid requires a Wobbe index of 37.8 – 46.8 MJ/Nm^3. Elsewhere, bio-methane produced in the Netherlands by one plant achieves the L-gas standard in that country with a typical Wobbe index of 44 MJ/Nm^3. This is similar to the natural gas supplied within the Netherlands by the Groningen gas field which has a low Wobbe index as a result of its relatively high nitrogen content.[181] However the gas grid in the UK requires the gas to be within the range of 47.2 – 51.41 MJ/Nm^3, and this is likely to result in a requirement for additional gas clean up.

[181] IEA (2006) *Biogas Upgrading to Vehicle Fuel Standards and Grid Injection; Task 37 – Energy from Biogas and Landfill Gas*

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

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

Approach Taken in This Study

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

Offset emissions for climate change impacts are based on the calorific value of natural gas, assumed to be 0.238 kg CO_2 per kWh (including emissions associated with extraction and transport). Biogas injected into the grid is also assumed to offset the air quality impacts associated with the pre-combustion emissions for natural gas such as those relating to transport and extraction. These are shown in Table C-14.

Table C-14: Pre-combustion Emissions Associated With the Use of Natural Gas

	Pre-combustion emissions, g / MJ
NMVOC	0.025
NOx	0.022
SOx	0.027
PM2.5	0.001

Source: ecoinvent (2004) ecoinvent Data v1.1, Final Reports ecoinvent 2000, No. 1-15, Swiss Centre for Life Cycle Inventories, Dubendorf, 2004

We assume similar environmental impacts associated with the gas clean up process as was the case where the biogas is upgraded for use as a vehicle fuel (outlined in Section C.4.2.1). Our analysis is based on data from German facilities where additional clean up steps (such as propane addition and oxygen removal) are not required if the gas is injected into a L-gas grid. However limited data is available from similar processes operating in other countries where such grids do not exist.

C.5 Monetised Benefits and Impacts Associated with the Solid Residue

C.5.1 Post Digestion Treatment Options

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

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

Whilst there may be some arguments for direct spreading, not least that of cost, it is considered better practice to stabilise the solid residues (following dewatering depending upon the materials and the process) through an aerobic stage so as to produce a compost. One of the reasons for stabilising the solids from digestion is to reduce the potential for nitrogen to leach following application – the stabilisation process makes for an amendment with high organic matter content, but with reduced availability of nitrogen. The level of nutrients such as phosphorous is much lower than in, say, sludge based materials, making it possible to apply more organic material without creating problems of groundwater pollution / surface water run-off.

Approach Taken in This Study

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

Additional emissions of CO_2 and NH_3 are assumed to occur in the aerobic phase, and some additional CH_4 is assumed to be emitted in the transition from the anaerobic to the aerobic phase. The use of the biofilter necessitates the use of some additional electricity.

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

C.5.2 Physical Characteristics

Within the Northern Ireland context, technology suppliers offered information on the characteristics of their output, and these are shown in Table C-15.

Table C-15: Nutrient and Physical Characteristics of Compost Derived from Digestate

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

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

Approach Taken in This Study

For the purposes of this study, we have assumed that the output material behaves in the same way as compost from the same feedstock produced through aerobic means. However, we have assumed a lower mass of compost produced of 300 kg per tonne of waste input. The nutrient content is deemed to be the same as in the case of composting of kitchen and garden waste.

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

C.5.3 Contaminants

The quality of solid residues from 4 different suppliers is shown in Table C-16. The one company who preferred to spread slurry on land did not give details of the content of the slurry.

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

Table C-16: Levels of Heavy Metal Contamination in Compost from Digestion Residue

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

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

C.6 Monetised Impacts

Monetised impacts associated with the use of compost are calculated on a similar basis to those attributed to the compost produced from in-vessel composting operations. Since some of the material has already been degraded by the AD process, there is less solid material to enter the post-digestion composting process, and therefore less compost will be produced (typically less than half of that produced from a similar quantity of material sent to an in-vessel composting process). 90% of the compost produced from AD facilities is assumed to be used in agriculture, with the remainder used for horticulture and amateur gardening (see Section B.3.2 for further discussion on this).

C.7 Non Monetised Impacts

A number of benefits and impacts associated with the use of compost have not been monetised, as was the case for the compost produced from in-vessel and windrow facilities. These are discussed in Section B.4.

D Landfill

D.1 Direct Emissions to Air from the Process

D.1.1 Climate Change Impacts

This section discusses landfill gas generation (related to the carbon content of the waste) and its capture. Assumptions regarding landfill gas management are summarised in at the end of the Section.

D.1.1.1 Landfill Gas Generation

In order to capture the relationship between degradation and residence time, our model links the nature of the constituent organic compounds to the release of greenhouse gases through time-dependent 'first order decay' functions, as is done in both the Land Quality Management (LQM) landfill model (used for inventorying the UK's greenhouse gas emissions for waste), and in the IPCC default model.[182] The equation is:

$$C_d = \sum_i C_i \times (1 - e^{-k_i t})$$

Where:

C_d = mass of carbon degraded;

C_i = mass of carbon in a given fraction;

k_i = the rate constant for a given fraction;

t = time in days.

Emissions of methane from landfill are allocated to specific years over a 150 year period. The degradation factors within the model have been validated to some extent through assessing the implied methane emissions from the materials and cross-checking against work undertaken in the United States and by the UK Environment Agency.[183]

The constituent carbon fractions degrade at different speeds as a result of variations in their chemical and physical structure. Our model uses three degradation speeds to represent the varying speeds at which carbon degrades within the landfill.[184] The simplified grouping of carbon fractions used within the model is shown in Table D-1.

Table D-1: Simplification of Carbon Fractions for Landfill

Speed of Decay	Carbon Fraction(s)
Fast	Sugars
Medium	Fats, Proteins, Cellulose
Slow	Lignin and some Cellulose*
Note: *Some cellulose is bound within the lignin and is therefore similarly resistant to	

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

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

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

degradation.

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

Table D-2 shows the impact of these differential degradation rates, and confirms the outputs of our model for various materials in terms of the proportion of carbon degraded - firstly after 50 years has elapsed, and then after a 150 year time period.

Table D-2: Degradation of Fractions in Landfill

Material	**Proportion of carbon degraded**	
	After 50 years	**After 150 years**
Food waste	83%	95%
Garden waste	71%	90%
Office paper	75%	89%
Newspaper	70%	85%
Textiles (natural fibres)	79%	88%
Wood	79%	92%
Note: Newspaper contains a greater proportion of lignin than other forms of paper hence lower degradation.		

Table D-3 shows outputs from our model with respect to the biological carbon content and methane generation of bio-waste in landfill.

Table D-3: Biological Carbon Content and Methane Generation of Bio-waste

		Food waste	**Garden waste**
Biological carbon content (kg carbon per tonne of fresh matter)	Total biological carbon	134	259
	Cellulose	40	153
	Lignin	12	80
	Protein	26	9
	Sugars	26	1
	Fats	29	0
kg CH_4 generated per tonne	CH_4 emitted after 50 years	96	158
	CH_4 emitted after 150 years	106	200
Notes: 65% of degraded biological carbon is assumed to form CH_4, a further 35% goes to CO_2. The table shows the total amount of CH_4 generated through the decay process. Landfill gas management will reduce the total amount of CH_4 emitted to the atmosphere through gas capture and oxidation (see Sections D.1.1.2 and D.1.1.4 respectively).			

To take account of the time profile of these emissions over the 150 year period, the damage costs for landfill emissions are discounted using a declining long-term discount rate as recommended in the UK Treasury's Green Book. Table D-4 shows the rates at which damage costs are discounted for the relevant time periods.

Table D-4: Declining Long-Term Discount Rate as Applied to Landfill Emission Damage Costs

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

Source: The UK Treasury Green Book

D.1.1.2 The Issue of Gas Capture

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

A previous assessment undertaken by Eunomia used a gas capture rate of 50%, an approach based upon two studies conducted on behalf of Defra by LQM and Enviros.[185] A study conducted by ERM on behalf of Defra, however, assumed a 75% capture rate over the 100 year timeframe assessed.[186] A subsequent ERM report acknowledged that if one moved the analysis beyond this (somewhat arbitrary) timeframe, lifetime capture rates might be around 59%.[187] Documentation supplied with the Golders model indicates that the expert review group formed as part of that study considered that 85% of the gas would be collected during the gas utilisation phases, and a lifetime 75% gas capture rate appears to have been suggested upon that basis.[188]

The wider literature suggests a range of estimates for the efficiency of gas collection with a distinction being made between instantaneous collection efficiencies and the proportion of gas that can be captured over the lifetime of the landfill. [189] Whilst instantaneous collection rates for permanently capped landfilled waste can be as high as 90%, capture rates may be much lower during the operating phase of the landfill (35%) or when the waste is capped with a temporary cover (65%).[190] In addition, gas collection is technologically impractical towards the end of the site's life. The Intergovernmental Panel

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

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

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

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

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

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

on Climate Change (IPCC) has recently stated that lifetime gas capture rates may be as low as 20%.[191] We would consider, however, that landfills in the UK are somewhat better engineered than in the general (global) case, although a recent report by the European Environment Agency uses the IPCC figure.[192]

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

D.1.1.3 Energy Generated From Landfill Gas

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

LQM carried out a survey of landfill operators to estimate the total flare capacity across UK landfills.[193] They noted within their analysis that:

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

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

D.1.1.4 Oxidation of Landfill Gas

Some of the uncaptured landfill gas will be oxidised as it passes through the cap to the surface, the proportion being dependent upon the nature of the cap. The US EPA suggests a range of 10% to 25%, with clay soils at the lower end of the range and top-soils being at the higher end. This reflects a figure proposed by Brown et al in 1999 in a study on behalf of what was then the DETR.[194] A similar value was proposed by the IPCC.

However, a recently published review of the wider literature on this subject suggests that the mean fraction of methane oxidised was 36% (an average across 42 studies taken in a

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[191] IPCC (2007) *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (Metz B, Davidson O R, Bosch PR, Dave R, and Meyer L A (eds)), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., pp 600

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

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

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

variety of locations).[195] We have retained the 10% figure assumed by the IPCC and US EPA, but acknowledge that this may overestimate fugitive emissions of methane occurring from landfill in many cases.

Approach Taken in This Study

For waste that has not been pre-treated, we assume that 50% of the landfill gas is captured by the landfill gas management system, and that a further 10% is oxidised through the cover of the landfill. Assumptions regarding the landfill of pre-treated wastes are outlined in Section F.

D.1.2 Air Quality Impacts

Whilst landfill gas is principally comprised of methane and carbon dioxide, approximately 1% of the volume of the gas is made up of trace elements. This can include up to 150 substances including halogenated organics, organo-sulphur compounds and aromatic hydrocarbons depending on the nature of the waste.[196]

The gases which are emitted in any one year are assumed to be related to the quantity of methane or CO_2 produced, depending upon whether one is considering raw gas or gas once combusted (Table D-5). 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:

- Where gas is flared, the emissions of other gases are calculated with reference to the studies by Enviros et al and White et al. The way this is done is by calculating the CO2 content of flared gas and calculating the emissions of other gases through the quantities relative to CO_2 as specified in the two studies mentioned;
- A similar approach is used to calculate fugitive emissions, but in this case, the other emissions are calculated relative to the calculated quantity of methane emissions; and
- For gas which is emitted from the gas engine, the emissions of other gases are calculated using the quantities estimated in other studies relative to calculated CO_2 emissions.

Table D-5: Non Greenhouse Gas Emissions to Air from Landfilling

	Emissions mg/Nm^3 landfill gas			Source
	Fugitive Ratio to CH_4	Flaring Ratio to CO_2	Generation Ratio to CO_2	
Methane	1	0.001818	0.005714	Enviros
Carbon dioxide	1.733333	1	1	Enviros
Carbon monoxide	3.03E-05	4.09E-04	4.09E-04	White et al
Hydrogen sulphide	4.66E-04	1.69E-08	1.69E-08	White et al
Hydrogen chloride	2.67E-06	8.64E-05	1.14E-05	Enviros
Hydrogen fluoride	5.33E-07	1.82E-05	1.14E-05	Enviros

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

[196] Komex (2002) Investigation of the Composition and Emissions of Trace Components in Landfill Gas, R&D Technical Report P1-438/TR for the Environment Agency, Bristol

Chlorinated HC	8.10E-05	5.10E-06	5.10E-06	Enviros
Dioxins and furans	0	3.36E-13	5.43E-13	Enviros
Total Particulates	0	3.64E-05	0.00002	Enviros
Nitrogen oxides	0	0.000455	0.002571	Enviros
Sulphur dioxide	0	0.000545	0.0002	Enviros
Cadmium	0	0	2.86E-07	Enviros
Chromium	7.12E-08	1.25E-08	1.25E-08	White et al
Lead	2.00E-08	2.49E-09	2.49E-09	White et al
Mercury	1.41E-08	2.49E-09	4.57E-09	Enviros
Zinc	1.68E-07	6.64E-11	6.64E-11	White et al
Nickel	0	0	3.71E-08	Enviros
Arsenic	0	0	4.57E-09	Enviros
Total VOCs	0.000333	7.73E-06	0	Enviros
Non-methane VOCs	0	8.64E-06	8.57E-05	Enviros
1,1-dichloroethane	0.000036	0	0	Enviros
Chloroethane	1.33E-05	0	0	Enviros
Chloroethene	1.47E-05	0	0	Enviros
Chlorobenzene	0.000032	0	0	Enviros
Tetrachloroethene	0.000044	3.64E-08	5.71E-07	Enviros
Poly-chlorinated biphenyls	0	0	0	White et al
Benzene	3.2E-06	0	0	Enviros

Source: Adapted from White P R, Franke M and Hindle P (1995) Integrated Solid Waste Management: A Lifecycle Inventory, Blackie Academic & Professional, Chapman and Hall; Enviros, University of Birmingham, RPA Ltd., Open University and Thurgood M (2004) Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes, Final Report to Defra, March 2004

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.

Landfills produce less of the pollutants for which dose response functions are tolerably well known. No external damage costs have therefore been developed for many of pollutants listed in Table D-5. These figures include impacts associated with the use of diesel at the facility, and a small amount of avoided emissions resulting from the generation of electricity from landfill gas.

D.2 Energy Used at Facilities

Landfills typically use both electricity and diesel. Diesel use is assumed to be 1.65 litres per tonne of material sent to landfill, whilst the electricity requirement is taken to be 1% of that generated. These assumptions are taken from a recent report produced in the UK by ERM which investigated the greenhouse gas emissions associated with a range of waste treatment facilities.[197]

D.3 Energy Generation

We assume that 60% of the captured methane is used for energy generation, with the remainder being flared. The gross efficiency of the gas engine at the landfill is assumed to be the same as that used to generate energy from biogas, i.e., 37%. The emissions assumed to be offset by this generation are dependent upon the energy mix of the country, discussed in Section A.4.2.

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

E Incineration

E.1 Direct Emissions to Air from the Process

E.1.1 Climate Change Impacts

E.1.1.1 Carbon Content of Waste Materials

Greenhouse gas emissions occurring as a result of the incineration of waste will be dependent upon the carbon content of the dry material, along with the overall efficiency of energy generation that results from the combustion of that material. Table E-1 details the carbon content of waste components together with their energy and moisture content.

Table E-1: Carbon Contents and Energy Content for Materials in the Waste Stream

	Total C (% fm)	Proportion of C that is non fossil	Energy content (lower heating value as received) MJ per kg	Typical moisture content
Paper	41%	100%	13	10%
Card	32%	100%	12	24%
Dense plastic	77%		35	10%
Plastic film	72%		33	15%
Textiles	49%	50%	15	19%
Glass	0%		0	2%
Ferrous metal	0%		0	3%
Non ferrous metal	0%		0	5%
Wood	32%	100%	12	30%
Garden waste	26%	100%	8	45%
Food waste	14%	100%	4	70%
Misc. combustibles	40%	50%	15	41%
Misc. non combustibles	7%		0	6%
Fines	30%	100%	5	41%

N_2O emissions are modelled based on previous research undertaken by Eunomia on behalf of WRAP.[198] The considerable uncertainty with respect to these emissions is acknowledged within the EU BREF note, which provided a range of 5.5 – 66 g N_2O per tonne of waste treated by the facility. We use the mid point of these values within the current analysis, and assume N_2O emissions to be 35.75 g of pollutant per tonne of waste to the facility. CH_4 emissions are negligible from incineration facilities.

E.1.2 Air Quality Impacts

Typical air pollution control (APC) technology installed in incinerators located in the majority of member states includes:

[198] Eunomia (2007) *Emissions of Nitrous Oxide from Waste Treatment Processes*, Report to WRAP, July 2007

- Bag filters, used to trap polluted dust (particulate matter) entrained with the exhaust gases;
- Semi dry flue gas scrubbing involving the use of lime neutralizes acidic pollutants (such as SOx) within the flue gas;
- Selective Non-Catalytic Reduction (SNCR) processes, used to thermally reduce NOx by injection of a reducing agent (ammonia or urea) into the post combustion flue gas;
- Activated carbon to deal with dioxin (and furan) formation.

Whilst SNCR processes typically allows the incinerator to meet the demands of the European Waste Incineration Directive with respect to NOx emissions, use of Selective Catalytic Reduction (SCR) techniques results in significant further reduction in NOx. SCR involves the addition of ammonia and the use of a catalyst (usually made of titanium oxide) to convert the NOx and ammonia into steam and nitrogen.[199] The reduction in NOx is typically achieved at the expense of additional energy expenditure.[200] The scientific literature also highlights the role of SCR in reducing the quantity of dioxins generated in the incineration process (i.e. the total quantity of PCDD/PCDF found in the emissions to air and in residues). In the calculated balance of dioxins from incineration, the emissions to air are relatively trivial relative to those found in residues.

In Germany, the Netherlands, Belgium and Austria, the use of selective catalytic reduction technology in incineration facilities is commonplace, resulting in a reduction in the emissions of SOx and NOx from such facilities. We have therefore reduced the direct air quality impacts for incinerators operating in these countries.

NOx emissions have a significant influence on the damage costs attributed to the air pollution from waste incineration facilities. Reductions in NOx emissions results in a considerable improvement in the performance of the facility with respect to external costs attributed to the non greenhouse gas air pollution impacts.

A significant proportion of the NOx emission from waste incineration is generated by the thermal process itself, and is not therefore directly linked to the nitrogen content of waste entering the facility. Data on the chemical constituents of waste varies considerably between different literature sources with the nitrogen content being particularly variable, largely reflecting the natural variation in the nitrogen content of organic material (likely to be the main source of nitrogen within residual waste). We have therefore based our assessment of the air pollution impacts of incinerators upon emissions data, rather than linking to specific chemical elements within the composition.

Approach Taken in This Study

Our analysis considers emissions from two types of incinerator:

1. A facility that meets the Waste Incineration Directive (WID), typical of those that have installed SNCR to reduce NOx emissions;
2. A facility that significantly out-performs the requirements of the WID through the installation of SCR and wet scrubbing techniques. Emissions are based on data obtained from plant operating in the Belgium with this type of equipment installed.

Emissions data for the facilities are detailed in Table E-2.

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[199] The ammonia is added to support the reduction reaction.

[200] To ensure the catalyst is not contaminated by other elements within the flue gas the SCR system is typically located just prior to the emissions stack. This requires the flue gas to be reheated using additional electrical energy.

Table E-2: Emissions from Incineration Facilities

	WID compliant facility		Significantly out-performs WID[2]	
	mg/Nm^3	g / t waste[1]	mg/Nm^3	g / t waste
PM10 / dust[3]	10.0	61.0	0.5	3.0
Dioxin (ng ITEQ/Nm^3)	0.1	0.0	0.0	0.0
NOx	200.0	1,220.0	45.0	274.5
SO_2	50.0	305.0	1.0	4.8
HF	10.0	6.1	0.0	0.0
HCl	1.0	61.0	0.5	1.2
CO	50.0	305.0	10.0	55.2
NMVOC	10.0	61.0	0.5	3.0
Total heavy metal	0.5	3.0	0.0	0.2

Notes:

1. Assumes an exhaust gas exit volume of 140 Nm^3/s, based on data provided by a 650,000 tonne per annum incinerator located in Paris (Source: ExternE)
2. Assumes the use of SCR and wet scrubbing techniques to reduce emissions
3. 70% of PM10 is assumed to be PM2.5 (Source: Chang et al)

Sources: Information Centre for Environmental Licensing (2002) Dutch Notes on BAT for the Incineration of Waste, Report for the Ministry of Housing, Spatial Planning and the Environment, The Netherlands, February 2002; European Commission (2006) Integrated Pollution Prevention and Control: Reference Document on Best Available Techniques for the Waste Treatment Industries, August 2006; ExternE (1999) Externalities of Energy, Vol 10: National Implementation, prepared by CIEMAT for the European Commission, Belgium; Chang M B, Huang C K, Wu J J, and Chang S H (2000) Characteristics of heavy metals on particles with different sizes from municipal solid waste incineration, Journal of Hazardous Materials 79(3): pp229-239

Air pollution control residues from waste incineration facilities consist of a mix of unspent reagents and chemicals extracted from the flue gas. They are typically treated as hazardous waste and are usually required to be sent to hazardous waste landfills. Chlorine, sulphur, heavy metals and dioxins are likely to be concentrated in the air pollution control residues produced by incinerators. Ironically, the better flue gas cleaning systems perform, the more likely it becomes that toxic materials are concentrated in these residues.

Several recent studies indicate that long-term impacts of landfilling this hazardous material may be significant. In a Dutch study comparing the costs and benefits of landfill with those of incineration, the environmental damages associated with air pollution control residues were considered as the most important externality associated with treatment in an incineration facility.[201]

One life-cycle study suggests:

> *'The evaluation of waste incineration technologies largely depends on the assessment of heavy metal emissions from landfills and the weighting of the*

[201] Dijkgraaf E and Vollebergh H (2004) Burn or Bury? A Social Cost Comparison of Final Waste Disposal Methods, *Ecological Economics*, 50: pp233-247

> *corresponding impacts at different points in time. Unfortunately, common LCA methods hardly consider spatial and temporal aspects.'*[202]

Using a geochemical model to model some pollutants, the same study concluded:

> *'Landfills might release heavy metals over very long time periods ranging from a few thousand years in the case of Cd to more than 100,000 years in the case of Cu. The dissolved concentrations in the leachate exceed the quality goals set by the Swiss water protection law (GSchV) by a factor of at least 50.'*

Since these impacts are only likely to be significant in the much longer term, they have been excluded from our model.

The principal determinant of air quality impacts associated with incineration facilities relates to the NOx emissions – both with respect to direct emissions from the treatment process and emissions generated through the use of diesel in the facility. However, a proportion of the impact is offset by NOx emissions avoided through the generation of electrical energy as was previously discussed.

E.2 Energy Used at Facilities

The energy usage of the plant depends upon the scale of plant, and the nature of the flue gas cleaning system. It also depends upon the presence or otherwise of:

- Mechanical pre-treatment systems;
- Incineration air preheating;
- Equipment for re-heating of flue gas;
- Waste water evaporation plant;
- Flue gas treatment systems with high pressure drops (which demand more powerful fans); and
- Changes in the energy content of input waste (necessitating use of fuel to maintain minimum combustion temperatures).

ERM's analysis suggests 3.9 kWh electricity is consumed per tonne of waste treated at an incinerator, with process diesel use indicated as 1.2 kg of per tonne of waste.[203] They arrived at these figures using Environment Agency data collected for the development of the waste model WRATE. However, they note in their report that:

> *These process data were used as a substitute for all thermal treatment processes. In reality the ancillary requirements of each will differ, but within the context of the research the more important parameter relates to the energy conversion efficiency of the process.*

ERM's energy consumption figures appear to be very low in comparison to values given in the wider literature. It is certainly true that the impacts associated with energy generation are far more significant than those associated with energy use, but this does not make the figures for energy use less important.

[202] Hellweg S (2000) *Time- and Site-Dependent Life-Cycle Assessment of Thermal Waste Treatment Processes,* Dissertation submitted to the Swiss Federal Institute of Technology

[203] ERM (2006) *Carbon Balances and Energy Impacts of the Management of UK Wastes*, Defra R&D Project WRT 237)

CEWEP's survey of 97 facilities during 2001-2004 suggested the average electricity used by incineration processes was 78 kWh per tonne of waste input.[204] The Draft BREF note for Incineration gives figures of:[205]

- Electricity use

  62 kWh per tonne – 257 kWh per tonne, average 142 kWh per tonne;

- Heat demand

  72 GJ thermal energy per tonne – 3,366 GJ thermal energy per tonne, average 433 GJ thermal energy per tonne.

These, in turn, are far higher than figures suggested in, for example, reports by Erichsen and Hauschild (46 kWh electricity per tonne) though this figure reflects only the operation of gas cleaning equipment.[206] The Flemish Institute for Technological Research (VITO) gave the following consumption of energy for processes with and without SCR (these were based upon incinerators operated by Seghers Better Technology):[207]

- Natural gas: 7.2 m3 per tonne
- Oil: 4 kg per tonne (or 4.7 litres per tonne)
- Electricity use (per tonne): 80 kWh with Selective Non-Catalytic Reduction (SNCR) pollution abatement, 85 kWh with Selective Catalytic Reduction (SCR) abatement.

To ensure the catalyst is not contaminated by other elements within the flue gas the SCR abatement system is typically located just prior to the emissions stack, which requires the 200°F flue gas to be reheated using additional electrical energy.[208]

Approach Taken in This Study

We use the CEWEP figure for electricity consumption with SNCR and VITO's figures for energy use assuming SCR within the current analysis. We have also used VITO's data for the diesel usage, but have assumed no natural gas is used by the process.

E.3 Energy Generation

The amount of electricity produced by an incineration facility can be calculated from the total calorific value of the input waste materials, multiplied by the electrical generation efficiency (usually expressed as a percentage). The efficiency of generation of electricity by an incinerator may be quoted gross, or net of any energy used in the plant itself. The energy use in the plant depends partly upon the nature of the flue gas cleaning system used, but also upon a range of other factors. The relationship to flue gas cleaning is important since it seems likely that as standards for abatement have improved, so the energy used in achieving those levels of abatement has increased also.

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[204] Riemann I (2006) *CEWEP Energy Report (Status 2001-2004): Results of Specific Data for Energy, Efficiency Rates and Coefficients, Plant Efficiency Factors and NCV of 97 European W-t-E Plants and Determination of the Main Energy Results*, updated July 2006

[205] A BREF note is a note prepared by the Joint Research Centre of the European Commission to give guidance to Member States as to what is implied by 'Best Available Techniques' under the Directive on Integrated Pollution Prevention and Control. European Commission (2005*) Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration*, Final Draft, May 2005

[206] Hanne L, Erichsen L and Hauschild M (2000) *Technical Data for Waste Incineration - Background for Modeling of Product Specific Emissions in a Life-cycle Assessment Context*, Elaborated as part of the EUREKA project EUROENVIRON 1296: LCAGAPS, sponsored by the Danish Agency for Industry and Trade, April 2000

[207] VITO (2000) *Vergelijking van Verwerkingsscenario's voor Restfractie van HHA en Niet-specifiek Categorie II Bedrijfsafval*, Final Report

[208] Note that this is not always the case – see European Commission (2005) *Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration, Final Draft*, May 2005

ERM suggested gross efficiencies of 20-27% for conventional incineration with steam cycle electricity generation in a recent report for Defra.[209] Fichtner quotes a 'realistic range' for net electrical efficiency of 19-27%.[210] The highest figures we have seen quoted are those quoted in the context of the Belvedere Inquiry where it was claimed that a net efficiency of 27% would be achieved. This was based around assumptions of a thermal efficiency of 84% and an electrical efficiency of 35%. These are optimistic in the context of efficiencies currently achieved and are likely to be deliverable only at large operating scales. The Draft BREF note gave no case where the net export of electricity exceeded 18%.[211] A survey of 25 incinerators across Europe generating electricity only reported a maximum *gross* energy efficiency of 27.9% with a weighted mean efficiency of 21.8% across the 25 facilities (the mean *net* efficiency was given as 17.7%).[212] The current analysis uses a *gross* efficiency of 27%, reflecting the top end of the range quoted by ERM and the CEWEP survey.

Whilst CEWEP supplies maximum values for heat and electricity generation for facilities operating in CHP mode, the survey data does not directly supply any information regarding the ratio of heat to electricity produced at each of the facilities concerned. Where thermal facilities are concerned, and where steam turbines are used to generate energy, there is a trade-off between the generation of electricity and the generation of heat.

In its submission to the DTI as part of a review of the Renewables Obligation, ILEX assumed electrical output would be reduced at an approximate rate of 1 MW of electrical energy for every 4 MW of heat off-take.[213] Data from CEWEP gives the maximum heat output from surveyed facilities surveyed producing only heat as 92.7%, suggesting a theoretical ratio of 3.3 MW heat for every MW of electricity.[214] The maximum heat output for any of the surveyed facilities operating in CHP mode was 83.9%, whilst the maximum electricity output for the CHP facilities was 26.9%. This suggests a ratio of 3.1 MW heat for every MW of electricity. However, the German Waste Incineration Association suggests that the ratio should be rather lower at 2.3 MW heat for each MW of electricity, based on the data from German facilities (the majority of which operate in CHP mode).[215] It is not clear, though, whether the German figures speak in terms of gross or net generation, or indeed, whether they take into account the heat load effect (in other words, these figures may relate to the electricity and heat actually put to a useful purpose).

The relationship between the electrical efficiency, heat generation efficiency and total generation efficiency (as outlined above) is shown graphically in Figure 13.

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[209] ERM (2006) *Carbon Balances and Energy Impacts of the Management of UK Wastes*, Defra R&D Project WRT 237)

[210] Fichtner Consulting Engineers Limited (2004) *The Viability Of Advanced Thermal Treatment Of MSW In The UK,* ESTET, March 2004

[211] European Commission (2005*) Integrated Pollution Prevention and Control, Draft Reference Document on the Best Available Techniques for Waste Incineration*, Final Draft, May 2005

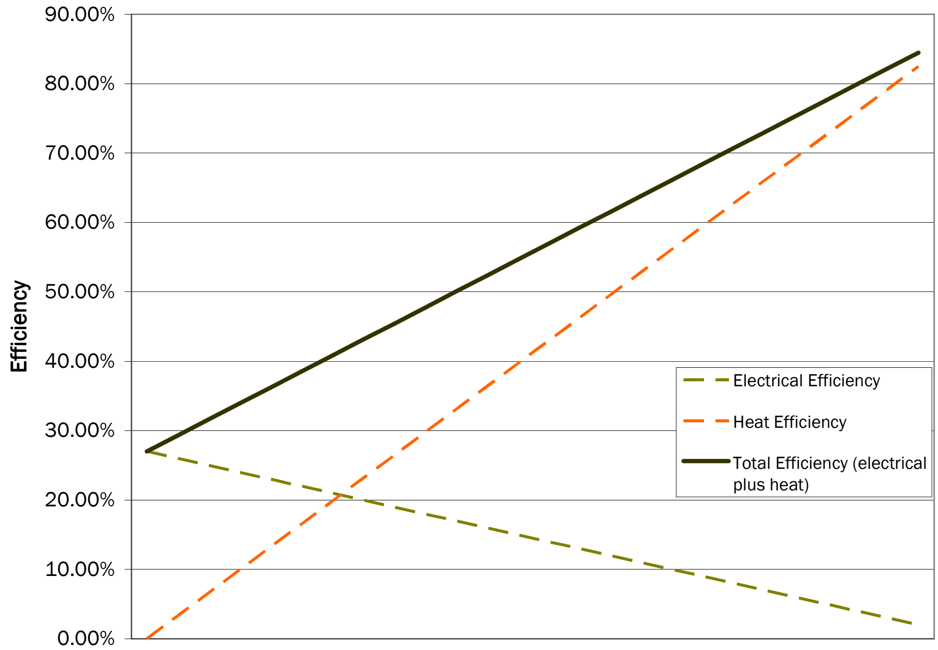
[212] Riemann I (2006) *CEWEP Energy Report (Status 2001-2004): Results of Specific Data for Energy, Efficiency Rates and Coefficients, Plant Efficiency Factors and NCV of 97 European W-t-E Plants and Determination of the Main Energy Results*, updated July 2006

[213] ILEX Energy Consulting (2005) Extending ROC Eligibility to Energy from Waste with CHP, Supplementary Report to the Department of Trade and Industry, September 2005

[214] This is simply calculated as the ratio of the maximum gross efficiency of heat generation relative to the maximum gross electrical generation efficiency of 29.7%

[215] Available from www.itad.de

Figure 13: Electricity, Heat and Total Efficiency – Facilities Operating in CHP Mode



Approach Taken in This Study

We have based our energy generation efficiencies on the survey data provided by CEWEP. Our energy generation efficiencies for facilities operating in CHP mode are based on the average electricity production for CHP facilities surveyed by CEWEP, using the higher ratio in the CEWEP report of 3.3 MW heat per MW electricity to calculate the heat production. We assume a total system generation efficiency of 66%, with electrical and heat generation efficiencies of 10% and 56% respectively. Our assumptions for all incineration facilities are summarised in Table E-3.

Table E-3: Summary of Energy Generation Efficiencies – Incineration Facilities

Parameter		**Assumption**
Electricity	Gross electrical generation efficiency	27%
CHP	Gross electrical generation efficiency	10%
	Gross heat generation efficiency	56%
Heat	Gross heat generation efficiency	85%
Notes 60% of the heat generated by incineration facilities is assumed to be utilised (in line with the central assumptions outlined in Section A.4.4.2).		

Food and garden waste are modelled with a lower heating value on an as received basis of 5.73 and 6.10 GJ / tonne respectively.

E.3.1.1 Climate Change Impacts

The amount of energy that is generated is related to the calorific value of the waste, and this will vary with the waste composition. The emissions offsets associated with this generation are, in turn, dependent upon the energy mix of the country for both electricity and heat. Green waste is assumed to generate 864 kWh of useful heat in a facility generating only heat. Offset greenhouse gas emissions associated with this electricity generation range from 180 – 279 kg CO_2 equivalent, depending upon the energy mix of the country. Facilities generating only electricity are assumed to generate 474 kWh of electricity, which results in avoided emissions of 9 – 474 kg CO_2 equivalent.

E.3.1.2 Air Quality Impacts

Whilst the incineration of waste results in direct emissions to air from the process (as previously discussed in Section E.1.2), the energy generated by the incinerator is also assumed to offset the air pollution that would have otherwise occurred from other means of generating that energy. The extent of this offset is dependent upon the energy mix of the country, as is the case with the avoided emissions associated with the climate change impacts. Thus the 864 kWh of heat energy generated by the combustion of green waste is assumed to offset NOx emissions of 312 – 589 g whilst the 474 kWh of electricity generated by combustion of the same material in an electricity only facility results in avoided NOx emissions of 35 – 1,248 g, depending upon the energy mix of the country.

E.4 Ash Recovery

We assume that 50% of the bottom ash from incineration is recovered for recycling. This, however, is not associated with any significant climate change or air quality impacts as the material is considered to be inert.

F Mechanical Biological Treatment

The term Mechanical Biological Treatment (MBT) encompasses many different types of technologies and combinations of treatment techniques. Our analysis considers three types of MBT facilities, all of which have an aerobic biological treatment element. These are considered to be representative of those most commonly operated across the different Member States. The processes modelled within our analysis are:

1. Aerobic stabilisation processes, where the output is landfilled after undergoing an aerobic degradation process to stabilise the waste;
2. Aerobic “whole waste” biodrying processes, where the aim is to dry the waste using the biological treatment phase and subsequently produce a fuel. Rejects from the process are stabilised prior to being landfilled;
3. Aerobic “splitting” processes, where the waste is split into low and high calorific fractions towards the start of the process. The high calorific fraction is thermally treated, whilst the low calorific fraction is stabilised prior to being landfilled.

These processes are described in more detail in the sections that follow.

F.1 Description of Processes

F.1.1 Aerobic Stabilisation Processes

The approach for modelling the impacts of stabilisation processes draws upon work by Eunomia on behalf of WRAP, which was based upon a raft of published research.[216] The body of research included work by Baky and Eriksson, Sonneson, and Komilis and Ham, all of whom investigated the link between the biochemical composition of the waste and the release of CO_2 within composting processes. This research, together with data sourced from technology suppliers, was used to model the degradation of carbon fractions within our model.

F.1.2 Aerobic Biodrying Processes

“Whole waste” biodrying systems involve the use of the heat from the process of biodegradation to reduce the moisture content of waste prior to its being mechanically refined (including using material separation technologies) for use as fuel. Examples of such systems include the Eco Deco process commonly used in Italy.

During this process degradation of some of the carbon fractions will occur, but the amount of degradation is relatively limited in comparison that occurring during aerobic decomposition (stabilisation) processes. Key differences between biodrying and stabilisation processes are the air-flow used to drive the process, the different water

[216] Schleiss K (1999) *Grüngutbewirtschaftung im Kanton Zürich aus betriebswirtschaftlicher und ökologischer Sicht: Situationsanalyse, Szenarioanalyse, ökonomische und ökologische Bewertung sowie Synthese mit MAUT*, Dissertation ETH No 13,746, 1999; Eunomia Research & Consulting, Scuola Agraria del Parco di Monza, HDRA Consultants, ZREU and LDK ECO on behalf of ECOTEC Research & Consulting (2002) *Economic Analysis of Options for Managing Biodegradable Municipal Waste*, Final Report to the European Commission; Komilis D P and Ham R K (2004) Life-Cycle Inventory of Municipal Solid Waste and Yard Waste Windrow Composting in the United States, *Journal of Environmental Engineering*, 130(11), pp.1390-1400; Baky A and Eriksson O (2003) Systems Analysis of Organic Waste Management in Denmark, *Environmental Project No. 822*, Copenhagen: Danish EPA; Sonesson U (1996) *Modelling of the Compost and Transport Process in the ORWARE Simulation Model*, Report 214, Swedish University of Agricultural Sciences (SLU), Department of Agricultural Engineering, Uppsala Sweden

management systems (in stabilisation processes, the waste is kept wet to maintain biodegradation, whilst in biodrying processes, the material is allowed to dry), and the retention times (which are much shorter in the case of biodrying). Biodrying processes are modelled using an analysis of data from technology suppliers.

The central aim of biodrying processes is to produce a solid recovered fuel (SRF) from the waste. A reject stream is also produced, which is assumed to be stabilised before being sent to landfill. The fuel is typically refined after biological treatment – removal of inert material such as stones and fragments of glass.

F.1.3 Aerobic Splitting Processes

Here the bulk of the residual waste is split prior to the biological treatment stage. The splitting process produces two fractions:

1. a low calorific fraction containing the much of the food waste and fragments of inert material, which is biologically treated (stabilised) prior to landfilling;
2. a high calorific fraction containing the bulk of the paper and plastics, which is combusted in a dedicated thermal facility or in a mass burn incinerator.

This process is typical of those used in Germany. These processes may produce a greater volume of fuel than the "whole waste" biodrying system. However the fuel may be lower in calorific value, as the material is likely to contain more moisture than that produced by the biodrying systems where the biological treatment is used to reduce the moisture content of the waste.

Direct emissions to air will result from the stabilisation of the low calorific fraction and the combustion of the SRF.

F.2 Direct Emissions to Air from the Process

Depending on the type of facility, direct emissions to air from MBT processes may result from:

- The Aerobic Stabilisation and Aerobic Biodrying processes;
- The incineration of SRF for those processes that produce a fuel;
- The landfilling of the stabilised residue or reject stream.

Emissions will result in impacts to both climate change and air quality. These are discussed in the sections that follow, for each of the different elements of the MBT processes.

F.2.1 Climate Change Impacts

F.2.1.1 Aerobic Stabilisation Phase

Principal impacts are biogenic CO_2 emissions resulting from the degradation of waste, resulting in 350 kg of CO_2 per tonne of waste to an aerobic stabilisation process. CH_4 emissions will be minimal in well managed facilities – typically in the order of 0.02 kg CH_4 per tonne of waste to the aerobic stabilisation process. N_2O emissions are similarly small (around 0.05 kg N_2O per tonne to facility).

Emissions impacts resulting from the aerobic stabilisation phase of the Splitting process will be less as a smaller volume of waste is treated by this part of the process. Biogenic

CO_2 emissions from the stabilisation phase in these process are typically 135 kg CO_2 per tonne of waste to the facility.

Emissions resulting from the stabilisation phase of the biodrying process are smaller still, both as a consequence of the smaller volume of waste, and the prior biological treatment phase. Typical stabilisation emissions from these types of processes are 80 kg of biogenic CO_2 per tonne of waste treated by the whole process.

F.2.1.2 Aerobic Biodrying Phase

Biogenic CO_2 emissions resulting from the biodrying treatment phase are smaller than those resulting from the aerobic stabilisation stage. The aim here is to dry the waste rather than to stabilise it, and as such the treatment phase is shorter, so that less of the carbon is released into the atmosphere as CO_2 gas. Emissions from this phase are in the order of 142 kg of biogenic CO_2 per tonne of waste to the facility. CH_4 emissions from the biodrying phase are typically 0.01 kg per tonne of input to the facility, whilst the N_2O emissions are 0.02 kg per tonne.

F.2.1.3 Combustion of SRF

Both the whole waste biodrying and splitting processes produce SRF which is assumed to be combusted in an incinerator. The whole waste biodrying process produces less fuel than the splitting process (typically 420 kg per tonne of waste to the biodrying facility compared to 529 kg for the latter type of process). Emissions from the incinerator are as discussed previously in Section E.1.1.

F.2.1.4 Landfill of Pre-treated Wastes

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

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

F.2.2 Air Quality Impacts

F.2.2.1 Aerobic Stabilisation Processes

The principal air quality impacts are emissions of NMVOCs and ammonia.

To minimise emissions from the stabilisation process, air circulation and/or a controlled air supply system is usually installed. Biofilters are typically used to reduce emissions of NMVOC and other organic pollutants (such as ammonia) emanating from the stabilisation process itself. These involve the use of microorganisms to biologically degrade the pollutants.

Most MBT processes are likely to use a minimum of a combination of ammonia scrubbing and biofilters to capture trace components from the exhaust gas produced during the stabilisation phase. MBT facilities in Germany use regenerative thermal oxidation (RTO) techniques to reduce these emissions. Although this results in greater emissions reductions, this must be offset against the additional energy expenditure which will in itself result in additional air quality impacts.

Optimised biofilters can remove 50-70% of the total organic content of the waste gas generated during these processes, although their efficiency is in part determined by the volume of gas produced.[217] Typical values for efficiency of compound removal are shown in Table F-1.

Table F-1: Biofilter Efficiencies of MBT Treatment Processes

Waste Gas Component / Substance Group	Biofilter Efficiency %
Aldehydes	75%
Alkanes	75%
Alcohols	90%
AOX	40%
Aromatic hydrocarbons (benzene)	40%
Aromatic hydrocarbons (toluene, xylene)	80%
NMVOC	83%
PAK, PCB, PCDD/F	40%
Odour	95-99%
Ammonia	90%

Source: Binner E (2002) The Impact of Mechanical-Biological Pre-treatment on the Landfill Behaviour of Solid Wastes, Biological Treatment of Biodegradable Waste: Technical Aspects, Workshop 8-10 April 2002, Brussels, pp355-372

Our analysis assumes the use of biofilters to reduce emissions of NMVOC and NH_3 to 95 g and 135 g of pollutant per tonne of waste to the aerobic stabilisation respectively. Emissions from the stabilisation stages of the biodrying and splitting processes will be correspondingly smaller as a result of the lower volume of material and previous biological treatment element of the biodrying process.

F.2.2.2 Biodrying Processes

As is the case with the aerobic stabilisation systems, biodrying plant typically install controlled air supply systems and use scrubbers and biofilters to reduce the impact of organic pollutants emanating from the aerobic degradation process.

Emissions of NMVOC and NH_3 will be lower as less degradation occurs. Typical emissions figures are 39 g of NMVOC and 53 g NH_3 per tonne of waste to the facility.

F.2.2.3 Combustion of SRF

Emissions from the combustion of SRF will be related to the quantity of fuel treated by this part of the process, as was the case for the climate change impacts. Our approach to modelling the air quality impacts associated with combustion in an incinerator has been previously outlined in Section E.1.2.

[217] Fricke K and Bidlingmaier W (2002) Gaseous and Sewage Emissions in Mechanical-Biological Rest Waste Treatment, *Biological Treatment of Biodegradable Waste: Technical Aspects,* Workshop 8-10 April 2002, Brussels, pp341-354

F.2.2.4 Landfill

Our model assumes the same emissions impact – per tonne of material to the landfill – for treated wastes as for untreated wastes (previously discussed in Section D.1.2). This is likely to overestimate the emissions impacts associated with landfilling this type of material, as it is clear that some of the volatile organic elements will have been degraded by the prior biological treatment phases. There is however little data upon which to estimate the extent of the likely emissions reductions. Air quality impacts for landfill are, however, relatively minor in comparison to the impacts resulting from greenhouse gas emissions.

F.3 Energy Used at Facilities

Energy is used in the mechanical and biological phases of MBT facilities for:

1. the removal of materials for refinement of fuel and recycling; and
2. shredding the waste.

Typically both diesel and electricity are used. There is an additional energy requirement where the output from the process is SRF which is subsequently combusted, as was previously discussed in Section E.2. Table F-2 summarises our assumptions for energy use during the different stages of the MBT process. The total amount of energy used will depend on the amount of material sent to each stage of the process. As such, the energy requirement for the incineration phase of the splitting MBT processes is higher than that of the biodrying process as a result of the larger volume of SRF that is produced by the former.

Table F-2: Energy Use by MBT Processes

Parameter	Energy used (per tonne treated by each phase)
STABILISATION PROCESS Electricity requirement Diesel use	 50 kWh 1.0 litre
BIODRYING / SPLITTING PROCESSES Electricity requirement Diesel use	 40 kWh 0.5 litre
INCINERATION Electricity demand for flue gas cleaning Diesel use by process	 78 kWh 4.7 litre

The energy used at the facilities will result in climate change and air quality impacts that are dependent upon the energy mix of each country, as was outlined in Section A.4.

F.4 Energy Generation

Our model assumes that the gross energy content – when considered on a MJ / kg basis - of the SRF produced from the biodrying process is higher than that produced by the splitting process, largely as a consequence of the reduction in moisture content. The splitting process produces a greater volume of fuel, however, and this results in greater energy generation overall per tonne of waste to the MBT facility. There is considerable variation within both types of process. In addition to process variations, the calorific value will also be affected by the composition of waste treated by the facility and this is likely to vary over time.

The typical gross energy content of SRF produced by whole waste biodrying process is 2,020 kWh, whilst that produced by the splitting process is 2,200 kWh per tonne of waste to the facility.

We assume that the SRF produced by the MBT processes is treated in an incinerator that generates only electricity. Our assumptions regarding the energy generation performance of incinerators are discussed in Section E.3. This energy generation will result in offset climate change and air quality impacts, as was outlined in the same section.