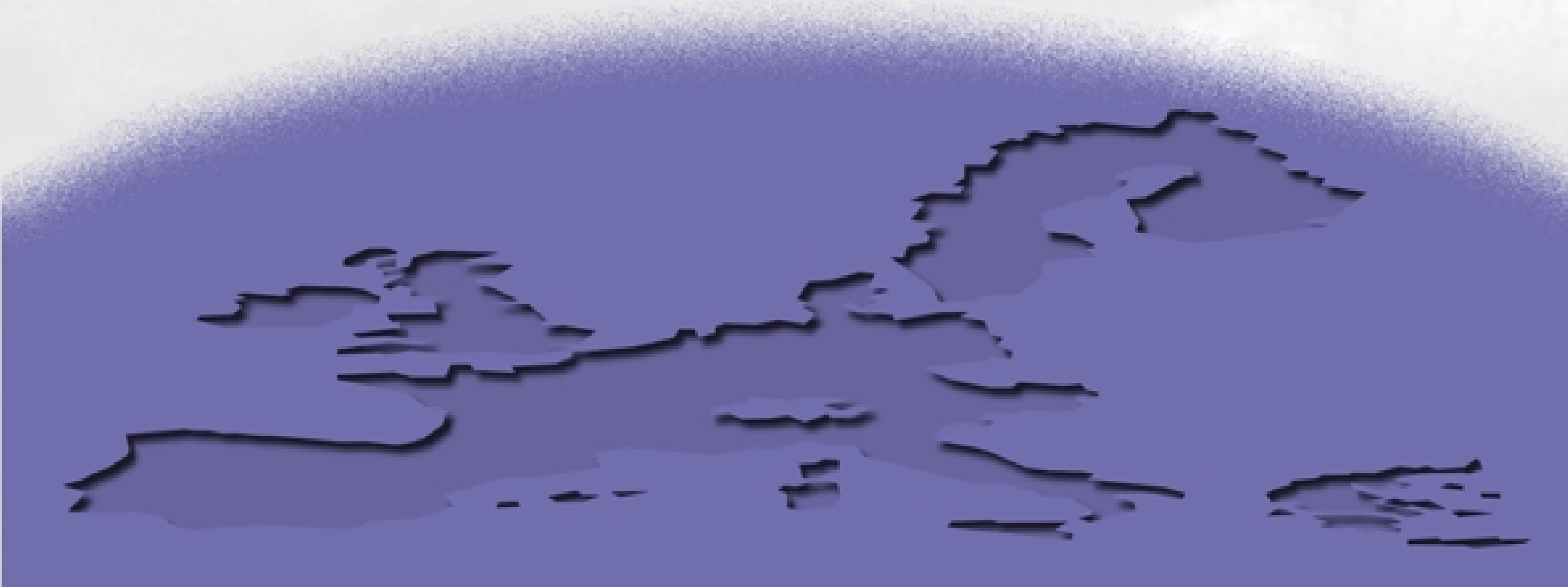
European Commission

Waste management options and climate change



Cataloguing data can be found at the end of this publication.

Luxembourg: Office for Official Publications of the European Communities, 2001

ISBN 92-894-1733-1

Waste Management Options and Climate Change

Final report to the European Commission,
DG Environment

Alison Smith
Keith Brown
Steve Ogilvie
Kathryn Rushton
Judith Bates

July 2001

AEA TECHNOLOGY
ENVIRONMENT

Title	Waste Management Options and Climate Change: Final Report
Customer	European Commission, DG Environment
Customer reference	B4-3040/99/136556/MAR/E3
Confidentiality, copyright and reproduction	This document has been prepared by AEA Technology plc in connection with a contract to supply goods and/or services and is submitted only on the basis of strict confidentiality. The contents must not be disclosed to third parties other than in accordance with the terms of the contract.
File reference	C:\0_WORK\PROJECTS\21158001\Final\Waste Management Options and Climate Change KB7.doc
Report number	Final Report ED21158R4.1
Report status	Issue 1.1

AEA Technology
Culham
Abingdon
OX14 3ED
Telephone +44 (0)1235 463134
Facsimile +44 (0)1235 463574

AEA Technology is the trading name of AEA Technology plc
AEA Technology is certificated to BS EN ISO9001:(1994)

Authors	Alison Smith, Keith Brown, Steve Ogilvie, Kathryn Rushton and Judith Bates

	Name	**Signature**	**Date**
Reviewed by	Keith Brown		
Approved by	Debbie Buckley-Golder		

Executive Summary

This document is the final report of a study undertaken for the European Commission Environment Directorate General by AEA Technology to assess the climate change impacts of options for municipal solid waste (MSW) management in the EU. The study covers the fifteen member states of the European Union and the time horizon 2000 to 2020.

The study is intended to inform developing EU-level waste policy, *in terms of climate change impacts only*. Climate change impacts are only one of a number of environmental impacts that derive from solid waste management options. Other impacts include health effects attributable to air pollutants such as NO_x, SO_2, dioxins and fine particles, emissions of ozone-depleting substances, contamination of water bodies, depletion of non-renewable resources, disamenity effects, noise, accidents etc. These environmental impacts are in addition to the socio-economic aspects of alternative ways of managing waste. All of these factors need to be properly considered in the determination of a balanced policy for sustainable waste management, of which the climate change elements are but one aspect. The study is not intended as a tool for municipal or regional waste planning, where local factors, such as the availability of existing waste management facilities and duration of waste management contracts, markets for recyclables, geographic and socio-economic factors, will exert the dominant influence.

The study assesses climate change impacts in terms of net fluxes of greenhouse gases from various combinations of options used for the management of MSW. The waste management options considered are:

- **Landfill of untreated waste.** Bulk untreated MSW is deposited in landfills. Alternative assumptions concerning the control of methane emissions in landfill gas (including the use of gas for electricity generation) are tested in the analysis.
- **Incineration.** Options assessed include mass-burn incineration of bulk MSW with and without energy recovery (as electricity only and combined heat and power - CHP), refuse-derived fuel combustion and pyrolysis and gasification;
- **Mechanical biological treatment (MBT)**. Bulk MSW, or residual wastes enriched in putrescible materials after the removal of dry recyclables, is subjected to a prolonged composting or digestion process which reduces the biodegradable materials to an inert, stabilised compost residue. The compost, which cannot be used in agriculture or horticulture because of its poor quality, is then landfilled. The treatment results in a significant reduction in methane forming potential of the compost in the landfill compared with untreated waste. Metals are recovered for recycling during the MBT process. Some of the paper and plastics in the incoming waste are diverted from the MBT process. These rejects are sent for either direct landfilling or incineration.
- **Composting.** Good quality garden and food wastes are segregated at source and composted, producing a bulk-reduced stabilised humus residue of compost that is of sufficient quality to be marketed as a soil conditioner or growing medium in agriculture or horticulture. Options of centralised composting facilities and home composting are considered.
- **Anaerobic Digestion** (AD). Like composting, this option produces a compost residue from source-segregated putrescible wastes for use in agriculture or horticulture. The

waste is digested in sealed vessels under air-less (anaerobic) conditions, during which a methane-rich biogas is produced. The biogas is collected and used as a fuel for electricity generation or CHP.

- **Recycling.** Paper, glass, metals, plastics, textiles and waste electrical and electronic equipment are recovered from the waste stream and reprocessed to make secondary materials.

Options are considered for MSW collected in bulk with limited recovery of recyclable materials and for materials segregated at source for more extensive recycling and (in the case of food and garden wastes) composting or AD. In addition to MSW, the study also assesses the greenhouse gas fluxes associated with managing waste electrical and electronic equipment (WEEE) disposed of with the MSW stream.

The principal processes quantified in the study that lead to *positive* greenhouse gas fluxes are as follows:

- Emissions of methane from the landfilling of biodegradable wastes (mainly paper and food and garden wastes – the latter known collectively as putrescible waste);
- Emissions of fossil-derived carbon dioxide from the combustion of plastics and some textiles in incinerators;
- Emissions of nitrous oxide during incineration of wastes;
- Emissions of fossil-derived carbon dioxide from the collection, transportation and processing of wastes, from the fuel used in these operations.
- Emissions of halogenated compounds with high global warming potentials used in WEEE (as refrigerants and insulating foam in fridges and freezers).

A number of processes lead to *negative* fluxes of greenhouse gases. These are as follows:

- Avoidance of emissions that would have been produced by other processes – for example:
  - energy recovered from incineration avoids the use of fossil fuels elsewhere in the energy system;
  - recycling avoids the emissions associated with producing materials recovered from the waste from primary resources;
  - use of compost avoids emissions associated with the use of any peat or fertiliser that it displaces.

- The study also takes account of non-fossil carbon stored (ie *sequestered)* in the earth's surface for longer than the 100-year time horizon for global warming adopted for the analysis. The main contributors to carbon sequestration are:
  - slowly degrading carbon stored in landfills receiving untreated biodegradable waste;
  - biodegradable waste stabilised by MBT treatment prior to landfilling, and
  - carbon in compost that is incorporated into stable humus in the soil

The net greenhouse gas flux from each waste management option is then assessed as the sum of the positive and negative fluxes. The study has also gathered information on the costs of alternative waste management options.

The **conclusions** are as follows:

1. The study has shown that overall, source segregation of MSW followed by recycling (for paper, metals, textiles and plastics) and composting /AD (for putrescible wastes) gives the lowest net flux of greenhouse gases, compared with other options for the treatment of bulk MSW. In comparison with landfilling untreated waste, composting / AD of putrescible wastes and recycling of paper produce the overall greatest reduction in net flux of greenhouse gases. The largest contribution to this effect is the avoidance of emissions from landfills as a result of recycling these materials. Diversion of putrescible wastes or paper to composting or recycling from landfills operated to EU-average gas management standards decreases the net greenhouse gas flux by about 260 to 470 kg CO_2 eq/tonne of MSW, depending on whether or not the negative flux credited to carbon sequestration is included.

2. The issue of carbon sequestration is a particularly important for landfills (and for MBT compost after landfilling), where the anaerobic conditions enhance the storage of carbon. Carbon sequestration plays a relatively small role in the overall greenhouse gas flux attributed to composting, because of the relatively rapid rate of decomposition of the compost after its application to (aerobic) soils.

3. The advantages of paper recycling and composting over landfilling depend on the efficiency with which the landfill is assumed to control landfill gas emissions. For sites with only limited gas collection, the benefits of paper recycling and composting are greater, but less when best practice gas control is implemented. In this case the net greenhouse gas savings from recycling and composting range from about 50 to 280 kg CO_2 eq/tonne MSW. If landfills further reduce methane emissions with a restoration layer to enhance methane oxidation, then recycling and composting incur a small net penalty, increasing net greenhouse gas fluxes to about 20-30 kg CO_2 eq/tonne MSW, if carbon sequestration is taken into account. If sequestration is neglected, then recycling and composting attract a net flux saving of about 50 (putrescibles) to 200 (paper) kg CO_2 eq/tonne MSW.

4. The study has also evaluated the treatment of contaminated putrescible waste using MBT, which may be appropriate if such waste cannot be obtained at high enough quality for composting with the aim of using the compost as a soil conditioner. MBT performed almost as well as AD with CHP in terms of net greenhouse gas flux from putrescible waste, but this advantage was largely determined by the credit for carbon sequestration. If this was not taken into account, then composting or AD of source-segregated wastes remained the best options. Omitting carbon sequestration significantly worsens the greenhouse gas fluxes calculated for landfills and MBT, but has a much smaller effect on composting or AD.

5. It must be emphasised that the apparent advantage of high-quality landfilling over composting and recycling of putrescibles and paper noted above refers only to greenhouse gas fluxes. Issues of resource use efficiency, avoided impacts due to paper making from virgin pulp and improvements in soil stability, fertility and moisture-retaining properties stemming from the use of compost in agriculture must all be considered as part of the assessment of the overall ‘best’ option. These factors are outside the remit of the present study, but their inclusion would almost certainly point to recycling and composting in preference to any form of landfill disposal for these waste components. Improving landfill gas management to reduce greenhouse gas emissions is

therefore essentially an ‘end of pipe’ solution, which reduces only one of the impacts of landfilling biodegradable waste without tackling the root cause.

6. For other materials (glass, plastics, ferrous metal, textiles and aluminium), recycling offers overall net greenhouse gas flux savings of between about 30 (for glass) and 95 (for aluminium) kg CO_2 eq/tonne MSW, compared with landfilling untreated waste. For these materials, the benefits are essentially independent of landfill standards and carbon sequestration.

7. For mainstream options for dealing with bulk MSW as pre-treatment for landfill, the option producing the lowest greenhouse gas flux (a negative flux of some 340 kg CO_2 eq/tonne MSW) is MBT (including metals recovery for recycling) with landfilling of the rejects and stabilised compost. MBT with incineration of rejects (energy recovered as electricity) gives a smaller net negative flux of about 230 kg CO_2 eq/tonne. Mass-burn incineration where half the plants operate in electricity only and half in CHP mode gives a net negative flux of about 180 kg CO_2 eq/tonne MSW. If all the incineration capacity were assumed to operate in CHP mode, then the net flux from incineration would be almost the same as from MBT with landfill of rejects. On the other hand energy recovery from incineration as electricity only would produce a net flux of only –10 kg CO_2 eq/tonne. These figures are based on EU-average landfill gas control, inclusion of carbon sequestered in MBT compost after landfilling and the replacement of electricity and heat from EU-average plant mix.

8. If the benefits of carbon sequestration are left out of the comparison of options just presented, then the MBT options both produce net positive greenhouse gas fluxes of 23 to 55 kg CO_2 eq/tonne MSW. Incineration is unaffected by assumptions on carbon sequestration.

9. The performance of MBT with landfilling of rejects is further improved as higher standards of landfill gas control are implemented, relative to mass-burn incineration, provided the contribution from carbon sequestration is included. If sequestration is omitted, incineration continues to perform better than MBT.

10. As stated in point 7 above, under the baseline assumptions used in this study, MBT with landfill of rejects gives rise to a lower (net negative) greenhouse gas flux than MBT with incineration of rejects. The main reason for this difference is lies in the source of greenhouse gas emissions in the two options. In MBT with landfill, methane emissions from the landfilled material is the main contributor to the positive flux, whilst for MBT with incineration, methane emissions are much lower but are more than outweighed by fossil carbon dioxide released from incinerating the plastic rejects. The relative performance of the two options depends crucially on the effectiveness of landfill gas control and, in the case of MBT with incineration, the energy source that is displaced by recovering energy from incineration. In the analysis performed here, we have assumed that electricity only is recovered, although in some cases there may be opportunities for recovering heat as well. This would further enhance the performance of MBT with incineration compared with MBT with landfill. It appears therefore that the choice between these options will largely depend on local circumstances, although either will offer a major improvement over current practices of landfilling untreated bulk MSW.

11. The issue of the source of displaced energy is critical to the performance of incineration in terms of net greenhouse gas flux. The base case is predicated on the assumption that energy from waste displaces electricity or heat generated at a CO_2 emission factor representative of average EU power and heat sources. For electricity, there has been an increasing trend to combined cycle gas turbine technology in recent years, but this has not been assessed separately because the emission factor from this technology is very close to average plant mix. Two alternatives to replacement of 'average' electricity are considered. They are (a) replacement of coal-fired power generation, and (b) replacement of electricity generated from renewable sources – in this case wind. The example given in (a) could come about, for example, from the accelerated retirement of an old coal-burning power station due to the commissioning of new incineration capacity, or through the use of RDF as a coal substitute. Example (b) may result from the inclusion of energy from waste (ie incineration) technology within a member state's target for renewable energy – as is the case in the UK. The greater the CO_2 emission factor of the replaced generation source, the greater the emission saved due to its replacement by incineration.

12. Replacement of coal-fired electricity generating plant by mass-burn incineration would result in a net negative greenhouse gas flux of almost 400 kg CO_2 eq/tonne MSW, with equal proportions of power only and CHP incineration capacity. Under these circumstances, mass-burn incineration would give practically the same emission saving as recycling and composting of source segregated materials. With all incinerators in CHP mode, mass-burn incineration would be the best overall option in terms of greenhouse gas flux. Combustion of RDF as a coal substitute in power stations or cement kilns gives rise to a net negative greenhouse gas flux of about half this sum.

13. A different picture emerges for the situation in which the electricity displaced by incineration comes from wind power, as an example of low-emissions renewable energy sources. Here the displaced generation source has almost no greenhouse gas emissions. In this case, mass-burn incineration is virtually neutral in greenhouse gas terms. In comparison, MBT with landfill of rejects produces a net negative flux of almost 340 kg CO_2 eq/tonne MSW, which makes it the best option for non-source segregated wastes. MBT with incineration of rejects gives a net negative flux of about 150 kg CO_2 eq/tonne MSW. These comparisons are on the basis of sequestered carbon being included in the overall flux from the MBT options.

14. If carbon sequestration is omitted, incineration and MBT with landfill of rejects have a similar net greenhouse gas flux in absolute terms (of 8 to 26 kg CO_2 eq/tonne MSW), whilst that for MBT with incineration is much higher, at about 135 kg CO_2 eq/tonne MSW.

15. Alternatives to mass-burn incineration have also been evaluated. From the perspective of greenhouse gas fluxes, emissions from pyrolysis and gasification are assessed as being similar to those of mass-burn incineration. Greenhouse gas fluxes from RDF manufacture and combustion (plus landfill of residues and recycling of recovered metals) depends highly on the fuel which they replace. Combustion as a replacement for average electricity plant mix results in higher greenhouse gas fluxes than for mass-burn incineration, due mostly to methane emissions from the landfilled residue left over from RDF manufacture. Improvements in landfill site gas control therefore improve the performance of this option relative to mass-burn incineration, although overall this RDF

option performs consistently worse in greenhouse gas flux than MBT with incineration of rejects.

16. Recycling of WEEE containing CFC refrigerants and foam agents now banned because of their ozone –depleting properties results in a net increase in greenhouse gas flux due to the escape of some of these agents during recycling operations. This leakage is more than sufficient to compensate for the considerable greenhouse gas benefits of recycling the metals from WEEE. Nevertheless, recycling of WEEE containing these materials is far preferable to landfill, where the greenhouse gas flux would be much higher. The use of less harmful refrigerants and foam agents and the adoption of more efficient collection procedures will largely eliminate the net positive greenhouse gas flux associated with WEEE recycling and result in substantial net greenhouse gas savings, due largely to the avoided emissions attributable to metal recycling. However, a considerable backlog of equipment containing CFCs remains to come through to the waste stream over the next 5-10 years and further efforts to minimise the release of GHG during recycling would be desirable.

17. Overall, emissions of greenhouse gas associated with transportation of waste, residues and recovered materials are small in comparison with the much larger greenhouse gas fluxes in the system, such as those related to avoided energy / materials, landfill gas emissions and carbon sequestration. Variations in emissions due to alternative assumptions about transport routes and modalities will therefore have a negligible impact on the overall greenhouse gas fluxes of the waste management options.

18. The study has evaluated four scenarios alternative scenarios of waste management in the year 2020 and compared the impacts on greenhouse gas fluxes with the year 2000. Achievement of the landfill directive's target to reduce the landfilling of untreated wastes in 2016 to 35% of 1995 levels is predicted to result in an overall reduction in greenhouse gas flux from a positive flux of 50 kg CO_2 eq/tonne in 2000 to a negative flux of almost 200 kg CO_2 /tonne in 2020. Even if achievement of the directive's target is delayed until 2020 (rather than 2016), then a negative flux of about 140 kg CO_2 eq/tonne results. Further reductions in greenhouse gas fluxes (to about –490 kg/CO_2 /tonne) could be achieved through investment in recycling, incineration with CHP and MBT. Alternatively, a scenario with no incineration and maximum biological treatment of waste achieves an overall greenhouse gas flux of –440 kg CO_2 eq/tonne.

19. The study has also examined the costs of waste disposal through the various waste management options, as reflected in disposal fees or the prices commanded by recycled materials. Wide difference in disposal costs exist between different member states. Landfill disposal, currently the cheapest option, will inevitably increase in cost with the requirement for higher environmental standards and the consumption of void space as existing sites fill up and close. Little information is available on the costs of MBT, but what there is suggests that this option may become increasingly competitive with landfill and incineration, especially when benefits of increased efficiency of landfill void space use and lower requirements for gas and leachate control are taken into account. Further growth in composting and AD for food and garden wastes will depend to a large extent on continuing success in reducing the costs of separate collection of feedstock and in establishing local markets for the compost product. Recycling remains highly dependent on the market value of the recycled product. With the principal exception of aluminium, the price of materials recovered from MSW does not cover the costs of

separating and reprocessing, compared with virgin materials, and such operations usually require subsidy. This is particularly so of plastic wastes. In this instance the option of co-incineration as a coal-replacement offers comparable greenhouse gas benefits to recycling but at a substantially lower cost.

20. Overall, the study finds that source-segregation of various waste components from MSW, followed by recycling or composting or AD of putrescibles offers the lowest net flux of greenhouse gases under assumed baseline conditions. Improved gas management at landfills can do much to reduce the greenhouse gas flux from the landfilling of bulk MSW, but this option remains essentially an ‘end of pipe’ solution. Incineration with energy recovery (especially as CHP) provides a net saving in greenhouse gas emissions from bulk MSW incineration, but the robustness of this option depends crucially on the energy source replaced. MBT offers significant advantages over landfilling of bulk MSW or contaminated putrescible wastes in terms of net greenhouse gas flux.

21. It must be emphasised that in practice other impacts of waste management options will need to be considered in addition to just greenhouse gas fluxes. These wider considerations will include factors such as resource use efficiency (which will, for example, impinge upon the choice between the *disposal* option of MBT and the *recycling* option of composting or AD) and the impacts of other emissions such as those associated with waste incineration. Furthermore, substantial environmental benefits are associated with the use of compost to improve soil organic matter status and more environmentally-benign methods of cultivation, but only the relatively modest benefits associated specifically with greenhouse gas fluxes have been considered in this study.

Acknowledgements

The project team wish to thank the numerous individuals and organisations who contributed information during the preparation of the draft report and to members of the expert review panel for their constructive and insightful comments during the finalisation of the study. The views expressed in this document are those of the authors and not necessarily those of the European Commission or any other body.

ABBREVIATIONS

AD	Anaerobic digestion
Al	aluminium metal
CCGT	Combined Cycle Gas Turbine
CF_4	Carbon tetrafluoride
CFC	Chloro Fluoro Carbons
CH_4	Methane
CHP	Combined Heat and Power
CO_2	Carbon dioxide
CV	Calorific Value
DDOC	Dissimilable Degradable Organic Carbon
DOC	Degradable Organic Carbon
EC	European Commission
EU	European Union
FBC	Fluidised Bed Combustor
Fe	iron (ferrous metals)
GCV	Gross Calorific Value
GHG	Greenhouse Gas
GJ	Giga Joule
GWP	Global Warming Potential (of greenhouse gases, relative to CO2, over a specified time horizon)
HCFC	Hydro Chloro Fluoro Carbon
HDPE	High Density Polyethylene
HFC	Hydro Fluoro Carbon
HHW	Household waste
IPCC	Intergovernmental Panel on Climate Change
kg	kilogram
kt	kilotonnes
kWh	kilowatt hour
LDPE	Low Density Polyethylene
LLDPE	Linear Low Density Polyethylene
MBT	Mechanical and Biological Treatment
MJ	Mega Joule
MRF	Materials Reprocessing Facility
MSW	Municipal Solid Waste
Mt	million tonnes
N_2O	Nitrous oxide
NCV	Net Calorific Value
OCGT	Open Cycle Gas Turbine
PE	Polyethylene
PET	Polyethylene terephthalate
PP	Polypropylene
PVC	Polyvinyl chloride
RCV	Refuse Collection Vehicle
RDF	Refuse Derived Fuel
RTS	Refuse Transfer Station
t	tonne
TJ	Tera Joule
WEEE	Waste Electrical and Electronic Equipment

Contents

1 Introduction

1.1 THE AIM OF THIS STUDY

The European Commission Environment Directorate General has contracted AEA Technology to undertake this study of the climate change impacts of options for managing municipal solid waste (MSW). The study covers the fifteen member states of the European Union (EU) and the time horizon 2000 to 2020.

The results will help to inform waste management policy at the EU level, but **only as far as greenhouse gas impacts are concerned**. Waste management has a wide variety of impacts on the environment apart from those associated with climate change and these impacts, which are outside the remit of the present study, but which also require proper consideration as part of a complete evaluation of the options. Some of the main environmental impacts of each waste management option are shown in Table 1. In addition, local factors exert a profound influence on the choice of waste management options, and for these reasons the output from this study are not aimed at informing waste management decisions at the *local* level.

Table 1 Some environmental impacts of the main waste management options

Option	Main environmental impacts
All options	• Emissions of carbon dioxide and other pollutants, noise, odour and congestion from vehicles transporting waste and by-products to and from treatment plants
Landfill	• Methane emissions from biodegradable waste, contributing to global warming and local hazards such as the risk of fires and explosions • Risks of water pollution from leachate (liquor) formed as waste decomposes • Land use – non-sustainable use of resources • Noise and odour • Some carbon compounds may be retained in the landfill for long periods (sequestered) and so not returned to the atmosphere as CO_2
Incineration	• Emissions of harmful airborne pollutants such as NO_x , SO_2 , HCl, fine particulates and dioxin • Emissions of carbon dioxide from fossil-derived waste (e.g. plastics) and N_2O contributing to global warming • Energy recovered can replace fossil fuels thus avoiding emissions of carbon dioxide • Fly ash and residues from air pollution control systems require stabilisation and disposal as hazardous waste • Bottom ash may be reused as a secondary aggregate - metals may be recovered for recycling from bottom ash
Recycling	• Saves energy (generally less energy is required to manufacture products from recycled feedstocks) and hence emissions of greenhouse gases and other pollutants • Prolongs reserves of finite resources (e.g. metal ores) – contributes to the sustainable use of resources • Avoids impacts associated with extraction of virgin feedstock (e.g. quarrying of ores and sand, felling of old growth forest to produce wood for paper)

Table 1 continued

Composting	• Avoids methane production from degradation of organic waste in landfills (as degradation is aerobic) • Compost can be used as a soil improver and can replace fertilisers and peat to some extent (both have negative environmental impacts) • Potential for carbon sequestration through increasing the store of soil organic matter • Improvements in soil fertility and soil organic matter content leading to possible down-stream benefits from reduced need for inorganic fertilisers, reduced need for irrigation and lower soil erosion rates. • Needs careful control of the composting process to avoid bioaerosols.
Anaerobic digestion	• As for composting, plus energy recovered can replace fossil fuels thus avoiding emissions of carbon dioxide
Mechanical biological treatment	• Reduces methane and leachate production from degradation of treated organic waste in landfills (as biological fraction is composted before disposal) • Materials may be recovered for recycling and/or energy recovery • More effective use of landfill void space since pre-treatment reduces bulk of waste needing disposal • Still dependent on landfill as repository of final waste, so not as sustainable as recycling or composting.

1.2 WASTE MANAGEMENT AND GREENHOUSE GASES

Human activity is increasing the concentration in the atmosphere of greenhouse gases. This is expected to result in a significant warming of the earth's surface and other associated changes in climate within the next few decades. The greenhouse gases that are making the largest contribution to global warming are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). All three are produced during the management and disposal of wastes. Estimated total emissions of these gases from the EU are shown in Table 2, which also shows the contributions from solid waste disposal. It should be noted that there is considerable uncertainty surrounding these emission estimates.

Table 2: Anthropogenic emissions of CO_2, CH_4 and N_2O in the EU in 1994 [1].

Direct GHG	Emissions (Mt)	GWP [2] (over 100 years)	Global Warming Equivalence of all emissions Mt equiv CO_2 (% from solid waste disposal)	Global warming equivalence emissions from waste disposal Mt equiv CO_2 (% of total waste management component for each gas)
CO_2 fossil	3,215	1	3,215 (<0.5 %)	15 (9 %)
CH_4	22	21	460 (33 %)	152 (89 %)
N_2O	1.05	310	325 (1 %)	3 (2 %)

Note: The global warming potential (GWP) is a factor that allows the concentrations of greenhouse gases to be expressed in terms of the amount of CO_2 that would have the same global warming impact. It depends on the spectral properties of the gas in question, its life time in the atmosphere and the time horizon chosen for climate change impacts. The GWP of CO_2 from fossil sources is assigned a value of unity. Methane and N_2O are, respectively, 21 and 310 times more potent in global warming terms than the same mass of CO_2 (over a 100-year horizon).

The impact of solid waste management on the global warming equivalence of European greenhouse gas emissions comes mostly from CH_4 released as biodegradable wastes decay under the airless (anaerobic) conditions in landfills. About a third of anthropogenic emissions of CH_4

in the EU can be attributed to this source [1]. In contrast, only 1% of N_2O emissions [3] and less than 0.5% of CO_2 emissions are associated with solid waste disposal.

For this reason it is often assumed that reducing the amount of CH_4 emitted from landfills would have the greatest potential for reducing the overall climate change impacts of solid waste management. Furthermore, because the atmospheric lifetime of CH_4 is relatively short (only 12 years), it is estimated that overall emissions would need to be reduced by about 8 % from current levels to stabilise CH_4 concentrations at today's levels. This is a much smaller percentage reduction than those needed to stabilise the concentrations of the other two major greenhouse gases, CO_2 and N_2O.

The developed countries have agreed under the UN Framework Convention on Climate Change (the Kyoto protocol) to reduce emissions of greenhouse gases [4]. For the EU, this amounts to a reduction on 1990 emissions of 8% in the period 2008-2012. Waste management policy will play a part in achieving this objective.

1.3 WASTE MANAGEMENT POLICY IN THE EU

Waste management policy in the EU enshrines the principles of sustainable development in the familiar waste management hierarchy, which underpins policy in this area. The hierarchy of waste management options places the greatest preference on waste prevention. Where wastes cannot be prevented, the order of preference decreases in order re-use, recycling, recovery of energy and finally (as the least preferred option) the disposal in landfills of stabilised wastes from which no further value can be recovered. With some 60% of MSW within the EU still being disposed of to landfill without any form of pre-treatment and extensive reliance on incineration for treatment of most of the remainder [10], it is clear that there is considerable scope for improvement.

As part of the suite of measures to improve the sustainability of waste management, the Landfill Directive (1999/31/EC) introduces requirements on member states to reduce the amount of biodegradable wastes disposed untreated to landfills. To achieve this objective, the Directive has introduced targets for reducing biodegradable waste disposed of to landfills to 75% of 1995 levels by 2006, reducing to 50 and 35% by 2009 and 2016[a]. The directive also requires improvements in environmental standards of landfills, in particular by requiring greater use of landfill gas collection and energy recovery from the methane in it, in order to reduce the main greenhouse gas impact of this waste management option.

To help meet the targets in the landfill directive, the European Commission is currently considering introducing further measures to encourage the adoption of alternatives to landfill for managing biodegradable wastes [5]. The general principles developed for the treatment of biodegradable wastes ('biowastes') are, in order of preference, as follows:

1. prevent or reduce biowaste production and its contamination by pollutants;
2. re-use biowastes (eg cardboard);
3. recycle separately-collected biowaste into original material (eg paper and cardboard) whenever environmentally justified;

---

[a] Member States which currently rely heavily on landfill have additional time to comply with these targets.

4. composting or anaerobic digestion of separately-collected biowaste that is not recycled into original materials, with the compost so produced being used in agriculture or for other environmentally beneficial purpose;
5. mechanical and biological treatment (MBT) of non-source separated biowaste as a pre-treatment for landfill disposal, and, finally;
6. use of biowaste for energy recovery.

To help inform the developing policy in this field, this study has undertaken a comparative assessment of the climate change impacts of landfilling biodegradable components in MSW and alternative treatments of recycling, composting, AD, MBT and incineration. The study focuses on emissions of greenhouse gases associated with the collection, transportation, treatment, use and disposal of materials arising from landfilling, recycling, composting and AD, MBT and incineration. It also considers the wider impacts of the waste management options in terms of greenhouse gas emissions elsewhere in the system. These are principally emissions averted by recovering energy from waste rather than using conventional fossil-based energy sources or through the use of recycled materials or compost in place of ‘virgin’ materials or peat/inorganic fertilisers. The study includes an assessment of waste management options for non-biowaste components in MSW (plastic, glass, metals etc) and waste electrical and electronic equipment (WEEE) that may enter the MSW stream. In addition to emissions of greenhouse gases from within the waste management systems and displaced emissions mentioned above, the study also estimates the scope for carbon storage (sequestration). Biogenic carbon that is sequestered for longer than the 100-year time frame for global warming is counted as a negative flux. This factor may be particularly important for carbon storage in landfills and in soil following the application of biowaste-derived compost.

The study focuses exclusively on the climate change impacts of waste management. It does not include any other environmental or health related factors (such as impacts on air, water or soil pollution, amenity impacts such as noise, odours and traffic and other accidents etc) that will also play a role in determining waste management policy. Whilst the focus of the study is on greenhouse gas fluxes, summary information on the private costs of waste management via the options assessed is also provided for comparative purposes.

The results from the study provide a comparison at the EU level between the waste management options for various waste components in terms of the greenhouse gas fluxes that drive climate change, indicating the distribution of emissions between the various steps in the waste management chain. Sensitivity analyses have been undertaken to assess the impacts of variations in key parameters on the overall greenhouse gas impact for each option. Finally, a scenario analysis is presented to compare three alternative views of waste management in 2020 with the overall position in 2000.

1.4 STRUCTURE OF THE REPORT

Further details on the approach and methodology of the analysis are given in section 2, which defines the scope of the analysis and provides a brief description of each waste management option addressed and the approach adopted in analysing the principal greenhouse gas impacts. Detailed information on each waste management options, including detailed background information, key assumptions, derivation of parameters and selection of values used in the analysis is then provided in a series of appendices. The reader may therefore refer to this detailed background information as required, without interrupting the flow of the main body of the report. Section 3 of the main report gives the results from the comparative analysis, along with the sensitivity analyses and an illustrative scenario analysis for the year 2020. The conclusions from the study are given in section 4.

2 Approach and methodology

This section outlines the approach and overall methodology used in the study. It defines the types of waste material and the waste management options considered and the climate change impacts that have been assessed. The section also provides an overview of how the climate change impacts for each waste management option were characterised for the analysis, which was undertaken using a spreadsheet model developed in Microsoft Excel 97 for Windows.

2.1 TYPE OF WASTE

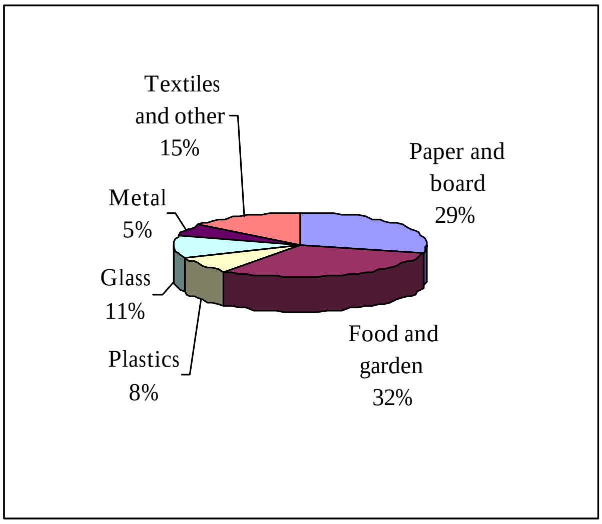
The study deals with management options for the various components of *municipal solid waste (MSW)*. Definitions of MSW vary from country to country, but the definition used in this study is that given by the landfill directive, namely:

> *'waste from households, as well as other waste which, because of its nature or composition, is similar*

This is compatible with the IPCC definition, which includes household waste, yard/garden waste and commercial/market waste.

According to the latest OECD data, total MSW arisings in the EU added up to about 170 million tonnes in the late 1990s (not all countries provided up to date information), the average composition of which is shown in Figure 1. In addition, the study also includes a comparison of options for dealing with waste electrical and electronic equipment (WEEE) produced by householders, since this too may have significant greenhouse gas impacts, although not all of this type of waste falls within the definition of MSW adopted above.

Figure 1: EU average MSW composition, based on OECD data for 1999 [10]



'Textile and other waste' is made up of 2% textiles, 6% miscellaneous combustibles, 2% miscellaneous non-combustibles and 5% fines (ie dust). The 'Metal' category is made up of 4% ferrous and 1% non-ferrous.. Food & garden waste is together known as 'putrescible' waste.

2.2 WASTE MANAGEMENT OPTIONS

Various options are available for the treatment of either whole MSW or of materials separated from it for recovery/recycling or pre-treatment prior to disposal. After waste prevention and re-use, the waste management hierarchy accords the highest preference to recycling, over energy recovery and disposal options. For economic success, recycled products need to find a market at a price that at least covers the cost of their recovery less any subsidies. The price commanded by recycled materials is highly dependent on quality, with clean, well-sorted and contaminant-free secondary materials commanding a higher price than mixed, low quality or dirty material. Indeed, in many instances low quality recyclate has no market and so must be disposed of at cost. Experience has shown that for MSW, segregation of material for recycling at the point at which it is produced (ie at households) provides the highest degree of clean, contaminant-free material for recycling. Two main types of waste management system are therefore considered, depending on whether bulk MSW or source-separation of various waste components is undertaken. The greenhouse gas fluxes associated segregating, collecting and transporting wastes are considered under the general heading of ‘mobilisation’, as follows

- **Mobilisation.** A common link between the waste management options is the need for collection, sorting, processing and transport from the source of the waste to the waste treatment / disposal facilities and end markets for recovered materials. All of these steps have greenhouse gas impacts, mostly through the use of fossil fuels and associated emissions of CO_2. As well as the direct transport of wastes and materials recovered from it, we also need to consider impacts due to residue disposal and any specific reagents required for the treatment option. Mobilisation processes and greenhouse gas fluxes are described in Appendix 1.

The waste management options considered in this study are outlined as follows:

2.2.1 Options for bulk collected MSW

- **Landfilling.** Landfilling involves the managed disposal of waste on land with little or no pre-treatment. Landfilling of biodegradable wastes results in the formation of landfill gas. The methane emitted in landfill gas is thought to represent the main greenhouse gas impact of MSW management. Currently about 60% of MSW in the EU is disposed of directly to landfills. As the least favoured option in the waste management hierarchy, landfill should be reserved for stabilised wastes from which no further value may be recovered. Landfill gas may be collected and either disposed of by flaring or used as a fuel. All components of MSW are currently acceptable for landfilling, including residual fractions left over after the separation of materials for recycling and the residues from pre-treatment processes such as incineration and MBT. Landfilling is described in detail in Appendix 2.

- **Incineration.** The most widely practised alternative to landfilling is mass-burn incineration, where bulk MSW is burnt with little or no pre-treatment. Modern MSW incinerators are required to recover energy released by the combustion process. Energy recovered from waste can replace the need for electricity and/or heat from other sources. The net climate change impacts of incineration depends on how much fossil-fuel carbon dioxide is released – both at the incinerator itself and in savings of fossil fuel from conventional energy sources displaced by incineration. The main residue from incineration is a volume-reduced inorganic ash, which has virtually no capacity to produce methane when disposed of in landfills. Incineration may therefore be considered as a landfill pre-treatment. Incinerators typically operate at scales of over 100,000 tonnes/year and require waste within a fairly narrow range of calorific value (CV). The disposal fees charged are supported by revenue from energy sales. With an operational life of 20-30 years, incinerators need a guaranteed supply of waste within specified composition ranges. Waste management planners must therefore take careful account of the impact of recycling activities on the availability and composition of waste destined for incineration under long-term contracts. For example, extensive recycling or paper or plastics could result in a residual waste enriched in food and garden wastes that would be too wet to incinerate. Incineration is detailed in Appendix 3.

There are several thermal treatment options available for refuse-derived fuel (RDF), made from paper, plastics and other combustible materials separated from bulk MSW. RDF may be burnt in dedicated combustors co-firing wood, peat or coal, or as a fuel supplement in cement kilns or coal-fired power stations. Although developed in the early 1980s, RDF technology is not widely deployed. Interest is also increasing in alternative thermal treatments such as gasification and pyrolysis. Here the waste is heated under carefully controlled conditions in either the complete absence of air, or with limited air supply, causing organic compounds to breakdown to form gaseous or liquid products that are then used to fuel engines to generate electricity. These options are not yet widely deployed for commercial scale MSW treatment.

- **Mechanical–Biological Treatment (MBT).** MBT is a pre-treatment option for landfilling. Raw MSW (or residual waste enriched in putrescible wastes after the removal of materials for recycling) is processed by a combination of mechanical and biological steps (shredding, sieving, composting and sometimes anaerobic digestion) to reduce the bulk and biological activity of the processed waste, which is then landfilled or used for landfill site cover or restoration. Recyclable or combustible materials may be removed from the waste for recycling or incineration. Pre-treatment of MSW by MBT prior to landfilling significantly reduces methane emissions from the landfilled waste, compared with untreated MSW. MBT is currently mostly confined to Austria and Germany. Appendix 4 provides a more detailed account of MBT.

2.2.2 Options for source segregated MSW

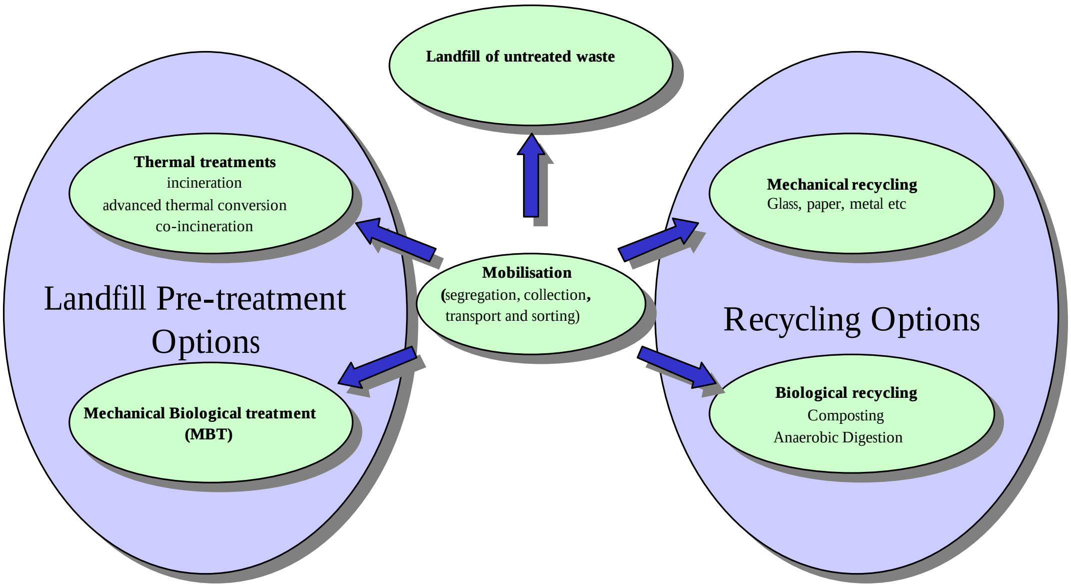
- **Composting**. Composting and the related process of anaerobic digestion (see below) are used for food and garden wastes. Composting makes use of micro-organisms to oxidise biodegradable wastes to carbon dioxide and water vapour, using oxygen in the air as the oxidising agent. A humus-like residue is left that is then used as a soil conditioner in agriculture or land reclamation or possibly as a growing medium in gardening or horticulture. Use of compost may have beneficial effects on greenhouse gas fluxes by replacing other products like fertiliser and peat and may also lead to increased storage of carbon in the soil (carbon sequestration). Industrial scale composting can be undertaken in open heaps that are turned and mixed mechanically (windrows), or alternatively in closed vessels with internal mixing and aeration. Composting can, of course, be undertaken with minimal equipment at home in most houses with suitable garden space. Efficient source-segregation of food and garden wastes destined for centralised composting is an absolute prerequisite if the resultant compost is to be of sufficient quality for marketing. See Appendix 5 for further information.

- **Anaerobic digestion (AD).** Like composting, AD is a biological process, but it takes place in sealed vessels in the complete absence of air (ie under anaerobic conditions). The process converts biodegradable waste to a biogas containing methane and carbon dioxide. The biogas is then used as a fuel, potentially displacing fossil-fuels. AD is essentially a controlled and accelerated decomposition process using the same types of micro-organisms that produce methane in landfills. The volume-reduced solid residue (digestate) is used like compost, but usually after a period of maturation by composting. Clean source-segregated feedstock is essential if the compost is to be suitable for marketing. See Appendix 6 for further information.

- **Recycling.** Recycling diverts components of the waste stream for reusing the materials contained within them. Provided the greenhouse gas impacts of separating and processing the recycled material into new products are less than those of manufacturing the products from primary material, then net saving results. Some materials can be recovered mechanically from bulk-collected MSW, such as metals recovered in incinerator ash and metals and glass recovered from MBT. The subsequent clean up of these materials for recycling is relatively straightforward and so there may be a market for them. To obtain higher quality of material requires segregation from other wastes at source. This is usually essential for paper and plastics recycling, and for all wastes, a higher price and better market access is usually achieved for source-segregated materials. Recycling is described in Appendix 7.

The waste management options are summarised in Figure 2.

Figure 2: Waste management options considered.



An option not explicitly considered in the present study is waste prevention. Waste prevention is invariably the most environmentally favourable waste management option. Not only does it avoid the need to process the waste itself, but it also eliminates the burdens associated with producing the material that becomes the waste in the first place.

A number of variations on each of the major options identified above are also evaluated in the study. These sub-options are listed in Table 3.

Table 3. Waste management options and their variations assessed in this study

Waste management option	Variations considered in the study
Landfilling of untreated wastes	• Landfill gas recovered and used for energy production • Landfill gas recovered and disposed of by flaring • No recovery of landfill gas
Incineration and other thermal treatments with energy recovery as electricity or electricity and heat. Metals recovered for recycling.	• mass-burn incineration with: - no energy recovery - energy recovered as electricity - energy recovered as heat and power (CHP) • pyrolysis/gasification • fluidised bed combustion of refuse-derived fuel (RDF) • co-incineration of RDF in cement kilns and power plants
MBT with metal recovery for recycling.	• with landfilling of reject fraction • with incineration of reject fraction
Composting with compost recovered for beneficial use in agriculture / horticulture.	• open systems (ie windrows) • closed systems (ie in-vessel composting) • home composting
Anaerobic digestion with compost recovered for beneficial use in agriculture / horticulture.	• with power generation • with heat and power recovery (CHP)
Recycling	• paper and cardboard • glass • plastics • iron and steel • aluminium • textiles • waste electrical and electronic equipment (WEEE)

2.3 SCOPE OF ASSESSMENT

The study evaluates greenhouse gas impacts and private financial costs of the waste management options listed above in the years 2000 and analyses four scenarios for waste management for the year 2020.

The model includes the following factors:

- Direct emissions from waste treatment processes
- Energy used (and hence greenhouse gas emissions arising) in the treatment and disposal of waste including transport
- Energy (and emissions) saved from reduced production of feedstock when feedstock is replaced by recycled materials (including replacement of peat or fertiliser use by compost)
- Energy (and emissions) saved from avoided transport of raw feedstock to the factory when recycled materials are used

- Energy process savings through the use of recycled feedstock
- Energy generation emissions avoided through energy recovery
- An estimate of emissions saved through the storage of carbon in landfill sites or in the soil following the application of compost.

The model does not include:
- Non-greenhouse gas impacts of waste management options
- Emissions from plant construction.

2.4 TREATMENT OF GREENHOUSE GASES IN THE MODEL

2.4.1 Carbon Dioxide and Methane

Carbon dioxide is released both during combustion of fossil fuels for energy used in waste treatment processes and directly from the waste during treatment. Carbon in the waste itself can be either released as CO_2 or CH_4 during the treatment process or remain in the waste or waste products (e.g. compost). These flows of carbon are shown in Figure 3 and Figure 4 for biodegradable and non-biodegradable materials respectively (for clarity, emissions of carbon from energy use or avoided emissions from energy production are not shown).

For biodegradable materials (putrescible waste comprising food and garden waste, paper and cardboard) the carbon will have been absorbed from the atmosphere by photosynthesis during plant growth relatively recently. If this carbon is released again as CO_2 during the treatment process then the carbon re-enters the natural carbon cycle. For this ‘short term’ carbon cycle, as the emissions have recently been offset by the uptake of an equivalent amount of carbon dioxide, then there is no net global warming impact, and no global warming potential is associated with the CO_2 emission, since the atmospheric concentration of short-cycle carbon dioxide is relatively constant from year to year. These emissions are reported here as ‘short term CO_2’ or ‘biogenic CO_2’ and are given a global warming potential of zero. If the emission occurs in the form of CH_4, (the atmospheric concentration of which has been rising as a result of man’s activities) however, then this has a higher global warming potential than CO_2, (Table 2) so must be accounted for.

In some organic materials, particularly plastics, the carbon originates from fossil carbon reserves laid down many millions of years ago. Reserves of these fossil fuels constitute an almost permanent sink for carbon. Combustion of fossil fuels releases the stored carbon into the atmosphere as fossil-derived CO_2, the concentration of which has almost doubled since industrialisation and is widely recognised as being the main driver for global warming. These emissions are reported as ‘fossil CO_2’ and have the usual CO_2 global warming potential of one. Fossil-derived organic materials in MSW are mostly plastics plus some textiles. They are essentially completely non-biodegradable[a] and the only way in which the carbon they contain may be released to the atmosphere as CO_2 is by combustion or other thermal processes.

---

[a] Some fossil-derived organic wastes, for example, certain household solvents such as acetone and industrial alcohol, are biodegradable, but as they form a tiny proportion of MSW, all the fossil-derived carbon is assumed to reside in non-biodegradable plastics.

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.

There is on-going debate as to whether this type of carbon sink will be included under the Kyoto Protocol. At present, the topic of carbon storage in soils is being considered for inclusion [6], but the issue of landfills as a carbon sink has not been raised. This study attempts to assess the possible size of such sinks. Given the uncertainties associated with the permanency of such sinks, we have examined the sensitivity of the results to our estimates of carbon sequestration potential.

Figure 3: Carbon flows in the management of biodegradable (short cycle C) wastes

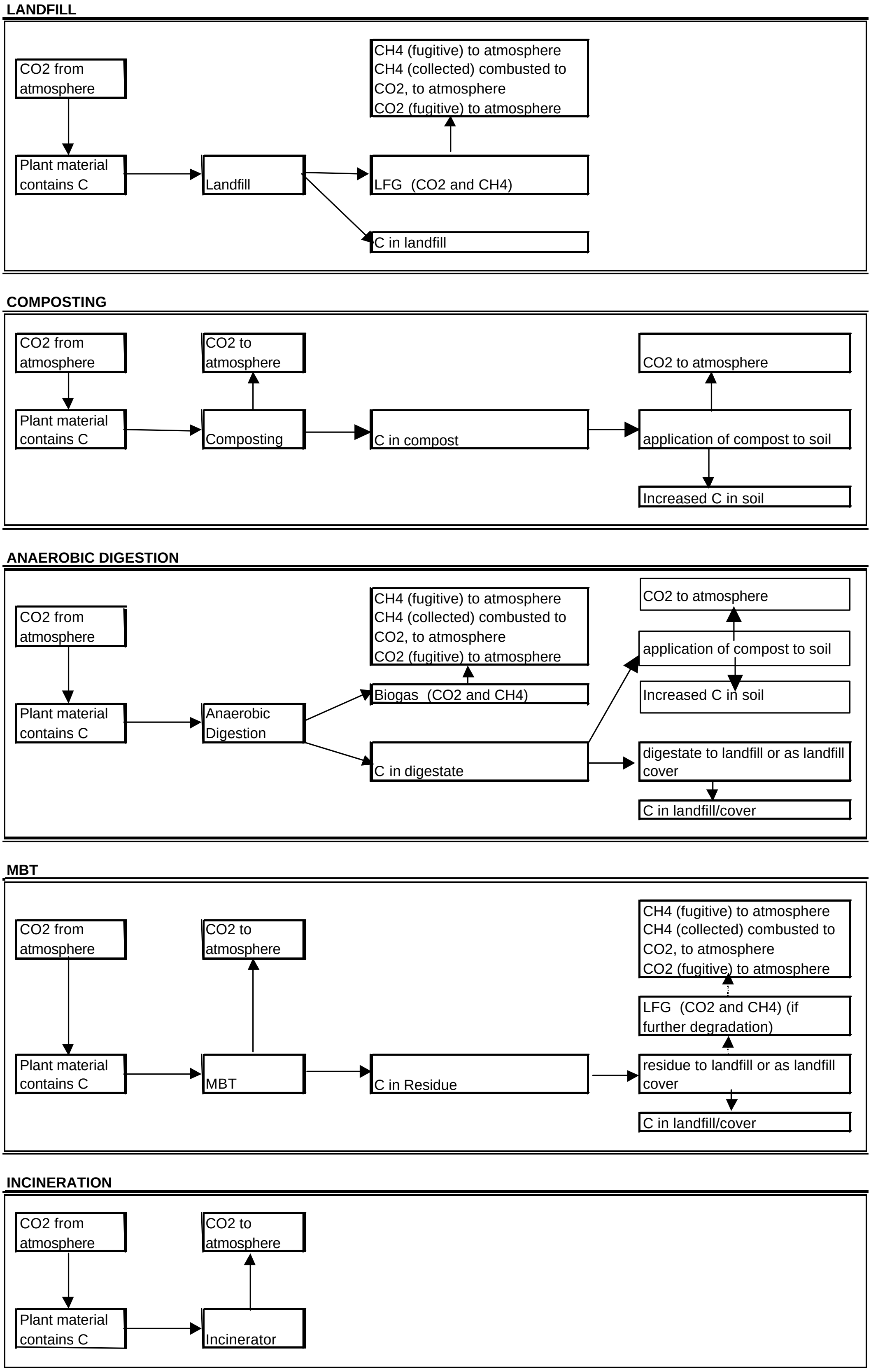
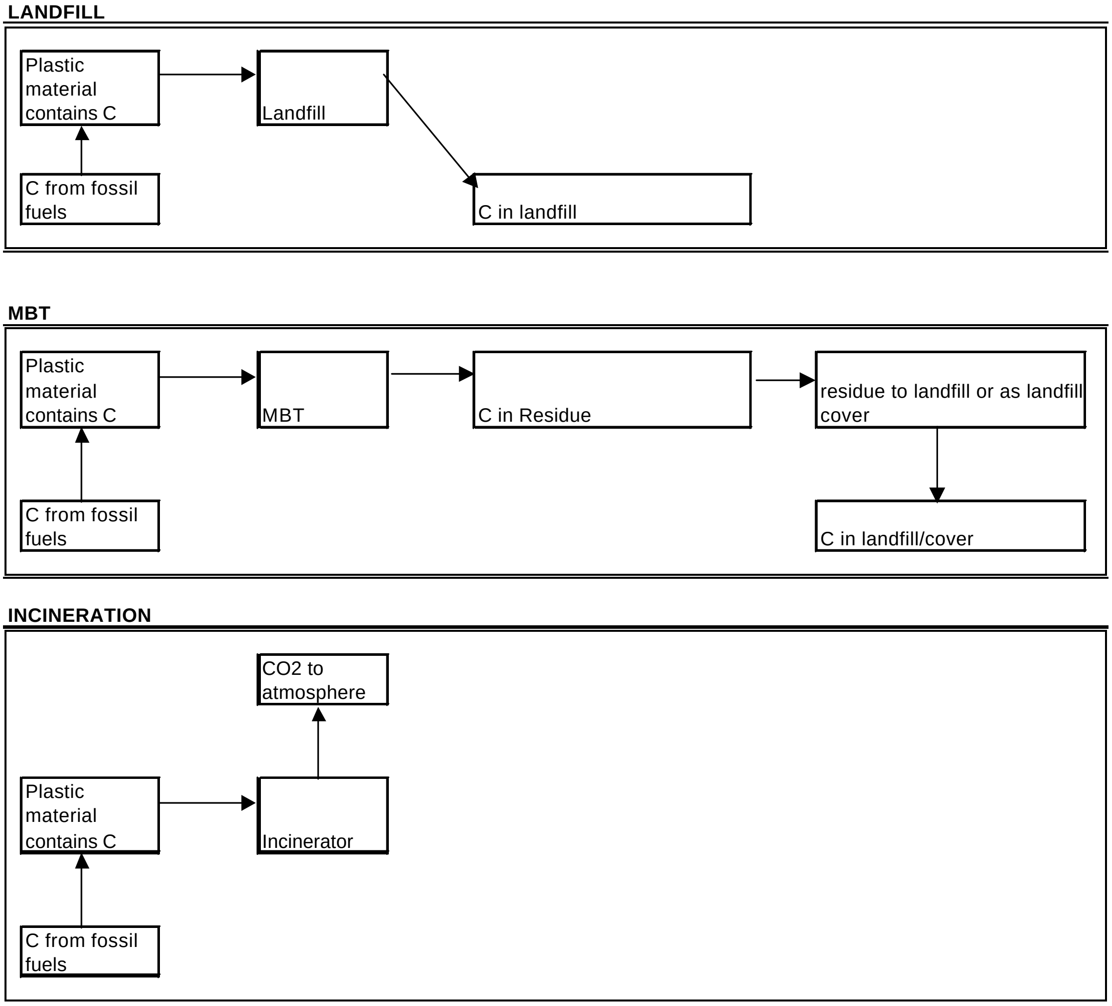


Figure 4: Carbon flows in the management of wastes containing fossil-derived (non-biodegradable) materials



2.4.2 Nitrous oxide and other greenhouse gases

The principal non-carbon greenhouse gas of interest to waste management is N_2O. Nitrous oxide is formed in trace amounts from nitrogen gas in the air and from compounds of the element present in waste during combustion in incinerators, landfill gas flares and combustion engines. Other sources of N_2O potentially relevant to this study include emissions from soil and fertiliser manufacture.

Other greenhouse gases that originate from waste disposal operations are the chloroflurocarbons (CFCs), originally used as aerosol propellants and refrigerants, and their replacements, HFCs and HCFCs[a]. These gases have very high global warming potentials. We therefore take account of the emission of CFCs, HFCs and HCFCs in the landfilling or recycling of household waste

---

[a] The use of CFCs is being phased out under international agreement under the 1994 Montreal Protocol, due to their ozone-depleting effect. HFCs and HCFCs are the main replacement at present. As refrigerators have service life times of 5-10 years, CFC-containing refrigerators will be coming through the waste stream for a several years to come. Aerosol cans formerly used CFCs as propellants, but these relatively short-lived products are rapidly declining in the MSW stream and so have been omitted from the present study.

electrical equipment, where fridges and freezers contain appreciable amounts that may be released into the atmosphere at the end of the product's life. The study also takes account of the displacement of emissions of carbon tetrafluoride (CF_4), a potent greenhouse gas used in primary aluminium refining, by recycling aluminium from MSW.

2.4.3 Global warming potentials and time effects

For consistency with standard practice in greenhouse gas assessments, all global warming potentials are those which apply over a 100 year time horizon [7]. Treatment of the main greenhouse gases is summarised in Table 4.

Table 4: Treatment of the principal greenhouse gases from waste management.

Emission	Origin	Trend in atmospheric concentration	Effective Global Warming Potential (100 years)	
			Emission	Carbon sequestration
CO_2 (fossil C)	Combustion of plastics	Increasing	+1	0
CO_2 (short-cycle)	Combustion and respiration of biomass C.	Stable	0	-1
CH_4 (short-cycle)	Methane-forming decomposition under anaerobic conditions	Increasing	+21	(not applicable)
N_2O	Combustion processes. Nitrogen metabolism in soils. Fertiliser manufacture.	Increasing	+310	(not applicable)
CFC-12	Refrigerant / insulation foam	Increasing	+8100	(not applicable)
CFC-11	Refrigerant / insulation foam	Increasing	+3800	(not applicable)
HFC-134a	Refrigerant / insulation foam	Increasing	+1300	(not applicable)
HCFC-141b	Refrigerant / insulation foam	Increasing	+600	(not applicable)
CF_4	Primary aluminium smelting	Increasing	+6500	(not applicable)

In line with the IPCC default methodology for waste, all greenhouse gas fluxes are treated as though they take place instantaneously. In fact, some fluxes such as emissions from landfills occur over a period of decades, and so the greenhouse impacts will vary with time. Here we assess total emissions, not their phasing. This simplification does not undermine the value of the approach in comparing waste management options in terms of overall greenhouse gas contributions. As greenhouse gas fluxes in this study have not been given an economic valuation, discounting is not required, and so the phasing of emissions within the 100-year time horizon can be ignored. Short-cycle carbon stored on land for longer than this time scale is considered to have been sequestered, and is so credited with a global warming potential of *minus* 1.

2.4.4 EU average approach

For many of the technologies assessed, emissions vary widely depending on the age of the plant, abatement technologies used, etc. Emissions will vary between the EU countries depending on technical differences and also differences in markets for secondary products such as compost or recycled goods. We have adopted an approach of looking first at 'EU average' emissions for the year 2000. 'Best practice' emissions will be lower than the 'EU average' emissions. The 'EU average' emissions will generally decrease in future years due to improvements in energy efficiency etc. We have attempted to take into account future emission changes when assessing future scenarios for the year 2020.

2.5 STEPS IN THE WASTE MANAGEMENT PROCESS

The waste management options assessed in this study impact upon climate change through a number of different steps. These fall into the following categories:

- **Mobilisation.** Climate change impacts of waste mobilisation are mostly indirect emissions associated with collection, sorting, processing and transporting waste. The main greenhouse gas is fossil derived carbon dioxide from vehicle fuels.
- **Process.** Process or treatment emissions include greenhouse gases derived from the waste itself (direct emissions) and from fuel used in its treatment (indirect emissions) prior to disposal of any residue. Examples of direct emissions include carbon dioxide emitted from waste combustion during incineration. Indirect emissions include those originating from fuel use in composting etc.
- **Disposal/use.** Greenhouse gas emissions result from the ultimate disposal of the waste in landfills or the use of materials derived from the waste. One of the main greenhouse gas impacts of waste management originates from methane emissions from biodegradable wastes in landfills. In addition, some short-cycled carbon is locked up in the landfills and prevented from being returned to the atmosphere as carbon dioxide for longer than the 100-year time horizon adopted in the study. This carbon is classed as being stored or sequestered. Waste management options in which sequestration is significant are landfill MBT and the use of compost from AD and composting plants.
- **Displaced emissions.** Emissions avoided as a result of useful energy or materials being recovered from waste displaces emissions that would have happened if alternative energy or materials had been used elsewhere in the system. When energy is recovered from waste, either as electricity, heat or both in combined heat and power (CHP applications), it displaces an equivalent amount of energy elsewhere in the system. The greenhouse gas emissions from this replaced energy recovery is therefore included in the analysis as an avoided emission. Waste management options which have an energy recovery component include incineration and other thermal treatments, AD and landfill where the gas is recovered for energy production. Recycling also displaces materials, together with their associated greenhouse gas impacts. This effect must also be taken into account, in terms of the net greenhouse gas impacts of making and using a product made from recycled material, compared with its alternatives. This issue is significant for recycled materials such as glass, metals and plastics, where recycling displaces the need to manufacture the product from virgin resources. It is also important in the case of waste-derived composts, which may replace inorganic fertilisers or peat in some applications.

The overall climate change impact of each waste management option is the sum of impacts for each of the stages listed above. The remaining part of this section provides an outline description of how each of these processes is treated in the analysis. Further details for each waste management option are given in the appendices.

2.6 MOBILISATION

Greenhouse gas impacts of mobilisation of waste for treatment include:

- Transportation of waste from point of arising to treatment facility, via any intermediate steps, such as household wastes sites, refuse transfer stations etc;
- Transport of residues and recycled materials from the waste treatment facility to ultimate disposal sites and markets.

Emissions are dominated by fossil carbon dioxide released by transportation. Nitrous oxide emissions account for less than 1% of the global warming impact of carbon dioxide emitted from vehicles [7], and so have been ignored. Mobilisation emissions were estimated from the emission characteristics of the vehicles employed in various stages of waste transportation and estimates of payload and average journey distances. Estimates range from about 4.2 to just over 12 kg CO_2 / tonne of waste, depending on the waste and option under consideration. Further details of the approach to estimating mobilisation emissions and costs are provided in Appendix 1.

2.7 PROCESS

Impacts from waste treatment considered in the analysis are summarised Table 5. Further information on the approach adopted for each impact is given in the appendices dealing with each waste management option.

Table 5: Greenhouse gas fluxes from waste management processes / treatments.

Impact	Approach adopted
Landfill	
• Fossil CO_2 emissions from fuel used in landfill operations.	• Estimated from fuel use per tonne of waste and emission factor.
• N_2O emissions from fuel used in landfilling operations.	• Negligible - not quantified.
Incineration & other thermal processes	
• Fossil CO_2 emissions from carbon compounds in the waste.	• Estimated from carbon content and origin in the waste.
• Short-cycle CO_2 from carbon compounds in the waste (no global warming impact).	• Estimated from carbon content and origin in the waste.
• N_2O emissions from waste combustion.	• Based on emission factors.
• CO_2 emission from fuel use for incineration.	• Internal use of energy is included in estimating power exported. This impacts on displaced emissions (see below).
MBT, composting, AD and recycling	
• CH_4 emissions during processing.	• Assumed to be zero for aerobic processes of MBT and composting. Fugitive emissions of 0.5% of CH_4 produced is assumed for AD.
• Short-cycle CO_2 from organic waste decomposition (no greenhouse gas impact).	• Estimated from data on organic matter degradation.
	• For composting and MBT, estimated from fuel use

• CO_2 from fuel used in treatment process.	per tonne of waste and emission factor. For AD, internal energy use is included in estimates of power exported and addressed under displaced emissions (below).
N_2O emissions from fuel used in treatment process	• Negligible - not quantified.

2.8 DISPOSAL / USE

Greenhouse gas impacts arising from disposal of waste components or use of waste derived products (such as compost) are summarised Table 6.

Table 6: Greenhouse gas fluxes from waste disposal / use of waste-derived products.

Impact	Approach adopted
Landfill	
• CH_4 emission in landfill gas. • Short-cycle CO_2 emission in landfill gas or in off-gas from flares or engines (no greenhouse gas impact). • Short-cycle carbon retained in the landfill for >100 years.	• Estimated using IPCC default methodology, based on estimates of degradable organic carbon content (DOC), proportion of DOC that dissimilates or mineralises (DDOC) and proportion of dissimilated carbon released as CH_4 or CO_2 during a 100-year period. The amount of CH_4 escaping to the atmosphere is estimated from the efficiency of gas collection for flaring or energy recovery and oxidation in the landfill. Non-dissimilated DOC is assumed to be stored for >100 years (ie is sequestered).
• Fossil CO_2 in landfill gas emissions.	• Not included – all fossil-derived organic compounds are assumed to be non-biodegradable.
• N_2O emissions from landfills or from landfill gas flares or engines.	• Not included – considered to be negligible.
• Release of CFCs/HFCs from WEEE	• Based on published burdens of CFC/HFC in WEEE and emission factors.
Incineration & other thermal processes	
• Emissions of greenhouse gases from thermal treatment residues (ash) after disposal.	• Not included – considered to be negligible.
MBT	
• CH_4 emission from MBT compost in landfill gas. • Short-cycle CO_2 emission from MBT residues landfilled or compost applied to soil (no greenhouse gas impact).	• Based on IPCC methodology for landfilled waste (see above), taking account of reduction in degradable carbon content during MBT treatment and impacts of gas control practices after landfill disposal.
• Short-cycle carbon retained in the landfill for >100 years.	• Non-dissimilated carbon is assumed to be sequestered.
• N_2O emission from landfilled MBT compost.	• Not included – considered to be negligible.
Composting and AD	
• CH_4 from compost applied to soil. Short-cycle CO_2 from compost applied to soil.	• Estimated from turnover time of organic matter added to soils.
• Short-cycle carbon retained in the soil for >100 years.	• Non-dissimilated carbon added to soils is assumed to be sequestered. Estimated from organic matter turnover time.
Recycling	
• Emissions of CFC/HFC during WEEE recycling.	• Based on published burdens of CFC/HFC in WEEE and emission factors.
• Other impacts from recycling	• No other significant disposal-related impacts associated with recycling.

2.9 DISPLACED EMISSIONS

Displaced emissions of greenhouse gases originate from the substitution of energy or materials derived from waste for alternative sources. We deal first with displaced energy emissions:

2.9.1 Displaced energy

The displaced energy components considered in the analysis are outlined in Table 7. The table is followed by further discussion of the approach adopted in deciding the source of the energy displaced by the waste management options.

Table 7: Displaced energy impacts

Impact	Approach adopted
Landfill	
• Energy recovery from landfill gas for electricity generation.	• Calculated from gas recovered, calorific value, conversion efficiency and assumed emission factors for displaced alternative energy source.
Incineration and other thermal processes • Electricity, heat and combined heat and power applications.	• Calculated from calorific values of waste components, conversion efficiency and assumed emission factors for displaced alternative energy source.
AD	
• Electricity, heat and combined heat and power applications.	• Calculated from gas recovered, calorific value, conversion efficiency and assumed emission factors for displaced alternative energy source.
Composting, MBT and recycling	• No energy recovered, so no displaced energy impacts.

A number of issues related to the estimation of the emissions actually displaced are common to all waste management options in where energy is recovered and these are addressed in the following paragraphs. Details concerning the calculation of energy recovered from each waste management option are given in the relevant appendices.

Energy recovered as part of the waste management process, either as heat, electricity or both in CHP applications, replaces the need for an equivalent amount of heat or power to be generated from other sources elsewhere in the energy system. The difficulty comes in knowing precisely what is being replaced. Some important factors affecting this assessment (for electricity generation) are:

- The **operating pattern** of the new generation system – peak lopping or base-load. Waste-based energy recovery systems such as incinerators or landfill gas schemes operate continuously. As part of an integrated power system, such schemes will compete with base-load generation. In much of the EU this would have been dominated by open-cycle coal or oil-fuelled steam turbine power stations but an increasing proportion is now provided by more efficient combined-cycle gas turbines (CCGTs). Renewable sources of energy (such as biomass and wind power) are also gaining ground. Nuclear power provides the mainstay of base-load power generation in France and Belgium. Nuclear power and renewable energy sources produce no direct greenhouse gas emissions, but have associated emissions from other parts of the fuel cycle. For example, emissions are produced during production of materials for construction of nuclear power plants, wind turbines, solar panels etc.

Emissions also arise from use of fuel and production of fertilisers for growing and harvesting biomass. However, emissions of greenhouse gases per unit of energy produced are much lower for renewable and nuclear sources than for fossil fuel sources.

- The **costs of generation** by the new source of energy compared with existing or proposed new alternative sources. Energy traded into the distribution system will, in a free market, compete on price with alternatives. Suppliers may enter into long-term contracts at negotiated prices to ensure that they have a guaranteed market for the energy recovered.

- **Subsidy schemes.** Governments may influence the market through taxes or subsidies to the advantage or disadvantage of particular energy sources. Governments often subsidise renewable energy in order to achieve environmental objectives. Some of these systems involve competitive bidding where different renewable options compete against each other, and the most cost-effective schemes are granted subsidies. This was the case with the UK government's NFFO (non-fossil fuel obligation) scheme. If energy from waste is included in these schemes, it is possible that it might displace other renewable energy schemes.

- The amount of **spare generating capacity** in the market. New plant coming on-stream may delay or prevent the need for commissioning new capacity elsewhere in the system, as opposed to displacing more costly existing older plant. New plant is likely to be CCGT or renewable energy (most countries are unlikely to commission new nuclear capacity for the time being, although Finland has just announced plans for a new plant).

- **Local factors** that may limit access to power or heat markets, and hence the extent of competition with alternative fuels. Local limitations on the grid may affect the nature of displaced generation. For example, island communities adopting waste incineration because of the shortage of a suitable landfill may have a completely different generation plant mix compared to the adjacent mainland, with a different spectrum of displaced emissions.

Wide differences in displaced emissions are therefore to be expected, depending on local circumstances. The range of typical CO_2 emission factors per unit of energy recovered is illustrated in Figure 5 (electricity generation) and Figure 6 (heat generation). The figures are based on the following information sources:

- The emission factor for the EU average mix was calculated from EUROSTAT 1998 figures for the total fuel mix for electricity production, with fuel use figures multiplied by carbon emission factors from the IPCC Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories.
- Emission factors for renewable sources were taken from the ExternE study [8,9].
- The emission factor for coal steam cycle plant assumed a generation efficiency of 36% and carbon content of coal of 25.8 t/TJ. The emission factor for CCGT plant assumed a generation efficiency of 50% and carbon content of gas of 15.3 t/TJ.
- The factors for gas and oil boilers assume heat generation efficiencies of 75% and carbon contents of 15.3 t/TJ (gas) and 21.1 t/TJ (oil).
- The EU-average industrial heat mix (excluding the iron and steel industry) was calculated using EUROSTAT 1995 fuel use data for industrial heat use (excluding iron and steel) and standard fuel carbon emission factors, assuming 75% boiler conversion efficiency.

Figure 5: CO_2 emission factors for electricity generating technologies

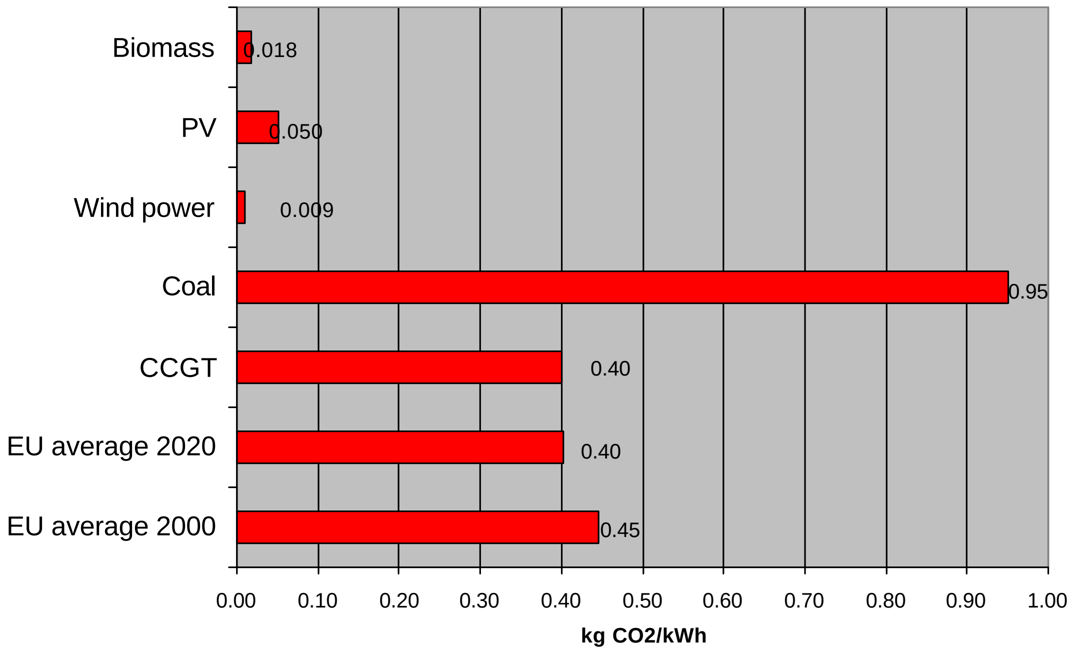
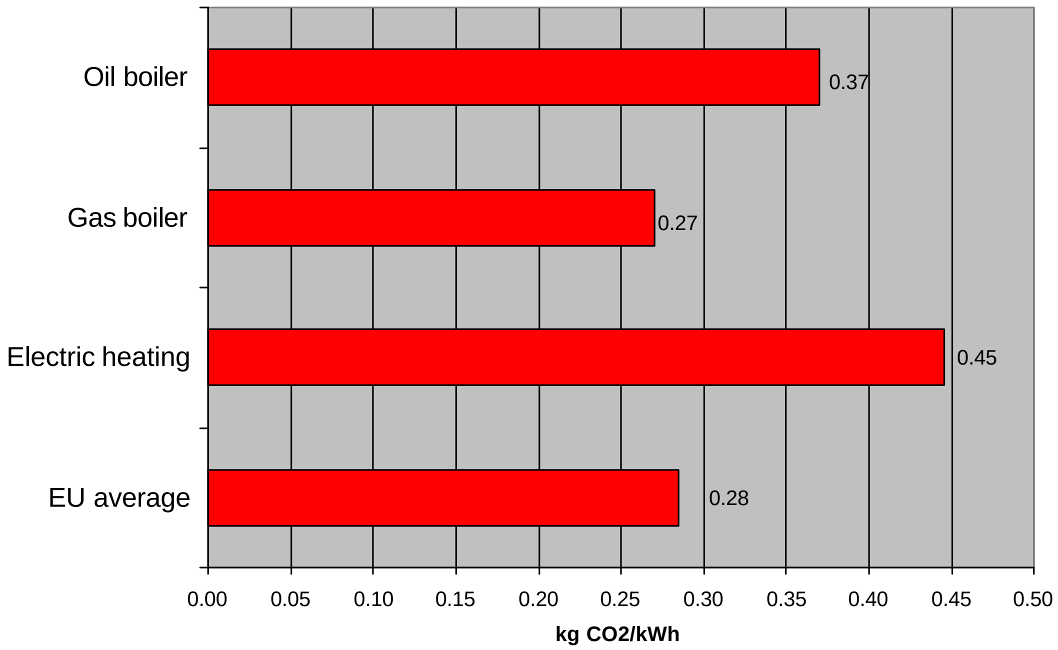


Figure 6: CO_2 emission factors for heat generating technologies



Common practice in life cycle assessment is to assume that either coal-fired steam cycle plant, CCGT or the average EU generating mix is displaced. For this study, we are also interested in the possibility of renewable sources being directly displaced by energy from waste schemes in cases where countries provide subsidy schemes where energy from waste competes on a cost basis with other renewable energy sources. **The baseline analysis assumes that the average EU generating mix is being displaced (this includes existing renewables, nuclear and CHP schemes as well as the conventional use of coal, lignite and CCGT). However, because of the critical impact of the source of the displaced energy for some**

technologies (notably incineration) which is sufficient to swing the outcome of the analysis, alternative cases in which wind or coal-generated electricity are also assessed. The CCGT case was not evaluated separately because emissions from CCGT are very similar to those from the current mix. Wind power was chosen to represent renewable energy because it has the lowest life cycle emissions (compared with solar photovoltaic power or biomass) and thus allows the lowest end of the sensitivity range to be assessed.

Heat recovered is considered to replace the EU-average industrial heat mix at 0.28 kg CO_2 /kWh (excluding the iron and steel industry, which uses a distinctive fuel mix not representative of the rest of EU industry) as shown in Figure 6. Alternative heat sources were not further evaluated in the study. It is, however, conceivable that a waste incinerator displacing electricity from biomass could also displace any useful heat from the same source. Heat-generating renewable energy sources are not widely deployed to input to heat mains suitable for incinerators in CHP applications, and so our choice of displacement of average heat technology emissions seems reasonable.

2.9.2 Displaced materials

Table 8 shows the greenhouse gas impacts associated with the displacement of alternative materials with waste-derived products are considered in the study. Details for each waste management option are found in the appendices.

Table 8: Climate change impacts through displaced materials.

Impact	Approach adopted
Landfill	No materials displaced by landfilling.
Incineration and other thermal processes	
• Ferrous metal recovery in incinerator residues.	• Assumed 90% of input ferrous metal is recovered, attracting the same displaced emissions (per tonne) as recycled material.
• Other recyclable materials recovered in incinerator residues (eg bottom ash).	• Not included.
MBT	
• Metals are recovered for recycling at the MBT plant.	• Assumed 100% of input metal is recovered, attracting the same displaced emissions (per tonne) as recycled material.
• Other recyclable materials recovered from MBT.	• Not included.
Compost & AD	
• Displaced fertiliser use due to nutrients in compost.	• Greenhouse gas emissions avoided by replacing mineral fertilisers, including fuel use for manufacture and N_2O emissions from N-fertiliser manufacture.
• Displaced use of alternative organic substrates.	• Reduction in fossil CO_2 emission from peat displaced by compost.
• Impacts due to changing farming practices associated with compost use.	• Impacts discussed but not quantified include changes in energy use for crop production and N_2O emissions from reduced input of N to soils.
Recycling	
• Impacts on greenhouse gas emissions from manufacturing products from recycled materials rather than virgin substrates.	• Fossil CO_2 emissions from fuel use in manufacture from primary materials as opposed to reprocessing recycled materials. Carbon tetrafluoride emissions averted by aluminium recycling.

2.10 COST ESTIMATION

The focus of this study is to quantify greenhouse gas fluxes from alternative waste management options, not to undertake a detailed cost-effectiveness analysis of the options, which would require a much more detailed analysis of economic costs. The costs data reported here are provided to give an indication of the likely costs of waste treatment and the range of variation observed.

DG Environment has requested the use of gate fees as a proxy for economic costs for this study. Gate fees are the fees charged by the operators of waste management facilities for disposal of received waste. They can be seen as representing the actual financial costs of waste management to the public more accurately than technology costs. Gate fees will be set at a level to recover all capital and operating costs, but will also include a profit element. In the case of recycling, sometimes a price is paid for receipt of recycled materials – this can be viewed as a negative gate fee.

In a competitive market, gate fees tend to be set at the level which the market can bear, and are therefore strongly influenced by the cost of nearby methods of waste disposal. For example, the fee charged by a composting plant may be set just below the fee charged by nearby landfill sites or incineration plants. Therefore gate fees vary very widely across the EU and also within countries. We have attempted to estimate average or typical values for each member state. Where this has not been possible, we have extrapolated gate fees from a member state with a similar geographical location, economic status and approach to waste management, as detailed in the text. Full details of the cost estimates are given for individual processes in the relevant appendices.

Gate fee estimates in national currencies and for years other than 1999 were converted to 1999 Euro using the appropriate year's conversion rate and the Eurostat Industrial Producer Price Index. As far as possible the level of any taxes, subsidies for plant, or premium prices for products have been indicated in the report.

The gate fee represents the cost of all treatment once the waste has been delivered to the treatment plant. However, costs are incurred prior to this in collecting the waste and, if necessary, sorting, cleaning, shredding or compressing it. These costs are normally borne by the local waste collection authority or by a private company which will recover the costs by sale of valuable scrap to the treatment plant. We have included estimates of these collection and pre-treatment costs as described in Appendix 1.

3 Results

This section presents the main findings of the study in terms of greenhouse gas fluxes from each waste management option and combinations of options. The details of each waste management options, including the mobilisation of the waste from point of arising to treatment centre (including the movement of any residues or reagents) is described in detail in the appendices. The appendices contain background assumptions and detailed tables of results. Key findings are presented graphically in this section, along with summary information on the characteristics of each option. Results are usually presented in terms of greenhouse gas flux per tonne of *MSW*, based on the average composition shown in Table 9. Information on fluxes per tonne of waste *component* are given in the appendices.

Table 9: Average EU MSW composition, derived mostly from OECD data. See Table 12 on page 55 for further details.

Component	Per cent fresh weight
Paper	29%
Putrescible	31%
Plastic	8%
Glass	11%
Metal	5%
Textiles	2%
Fines	5%
Miscellaneous. combustible	6%
Miscellaneous non combustibles	2%

The following topics are addressed:

- Options for bulk (non-source segregated) MSW
- Options for source-segregated MSW
- Waste Electrical and Electronic Equipment that enters the MSW stream, and
- Disposal costs

The final part of this section is comprised of a discussion of the key uncertainties, followed by an analysis of alternative waste management scenarios for 2020. Conclusions from the analysis are presented in section 4.

3.1 OPTIONS FOR NON-SOURCE SEGREGATED WASTES

There are four main waste management options for dealing with bulk MSW – ie waste that is collected in bulk from households, with any segregation for recycling and recovery of particular materials undertaken at a MRF or at the waste treatment/disposal facility. These options are:

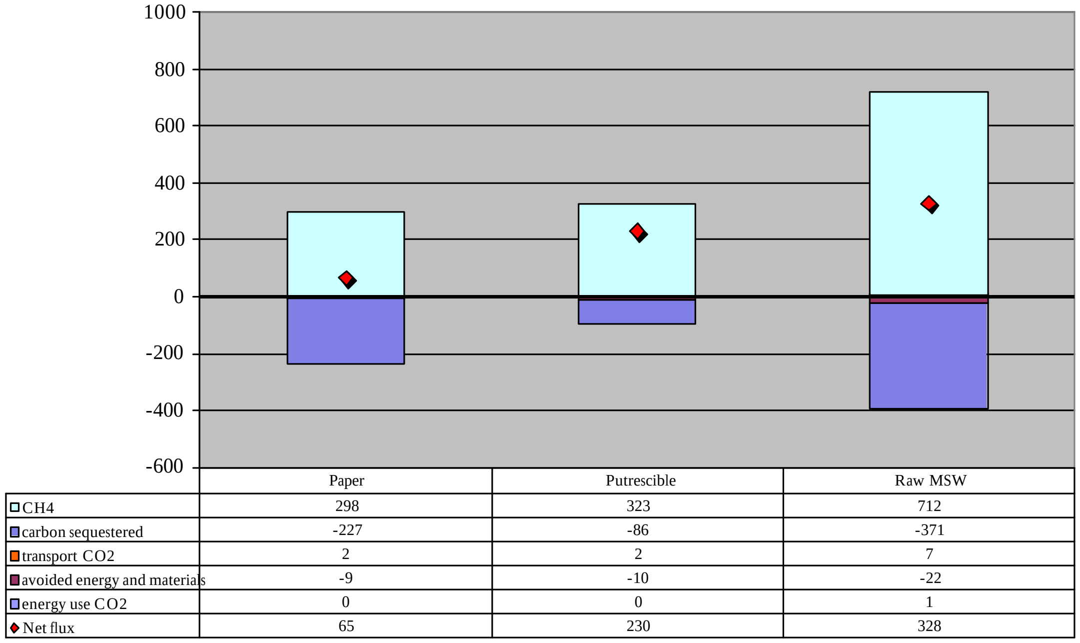
- **Landfill** – waste is collected in bulk and taken to the landfill. No materials are recovered or diverted.

- **Mass-burn incineration** – bulk waste is taken to an incinerator. Ferrous metal is recovered for recycling from the ash, which is then disposed of to landfill.
- **RDF combustion** – bulk MSW is collected and taken to a MRF, where plastic, paper and textiles are converted to RDF for combustion. The residual waste is disposed of to landfill, after removal of metals for recycling.
- **MBT** – bulk MSW is processed by MBT with various fractions being removed for recycling, incineration or direct landfilling. The composted stabilised residue is then landfilled.

3.1.1 Landfill

Detailed information on landfill is given in Appendix 2. Greenhouse gas fluxes associated with landfilling paper, putrescible wastes and raw MSW(expressed in terms of CO_2 eq/tonne of MSW) are shown in Figure 7. The graph refers to base case assumptions concerning the amount of degradable carbon that dissimilates to methane and CO_2, (ie DDOC content) and assumes EU-average landfill gas collection efficiency. Greenhouse gas fluxes are dominated by methane emissions (positive) and sequestered carbon (negative). The category labelled 'avoided energy and materials' refers to electricity generated from landfill gas. All other flux sources are negligible for the landfill option.

Figure 7: Greenhouse gas fluxes due to landfilling untreated MSW, paper and putrescible waste (kg CO2 eq / tonne MSW) – base case dissimilable carbon content and EU-average gas collection.

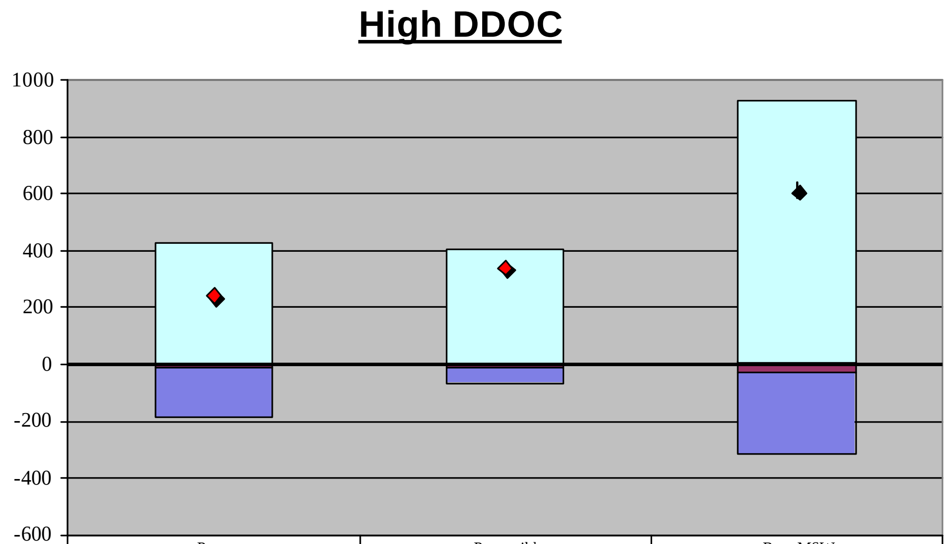


	Paper	Putrescible	Raw MSW
□ CH4	298	323	712
■ carbon sequestered	-227	-86	-371
■ transport CO2	2	2	7
■ avoided energy and materials	-9	-10	-22
■ energy use CO2	0	0	1
◆ Net flux	65	230	328

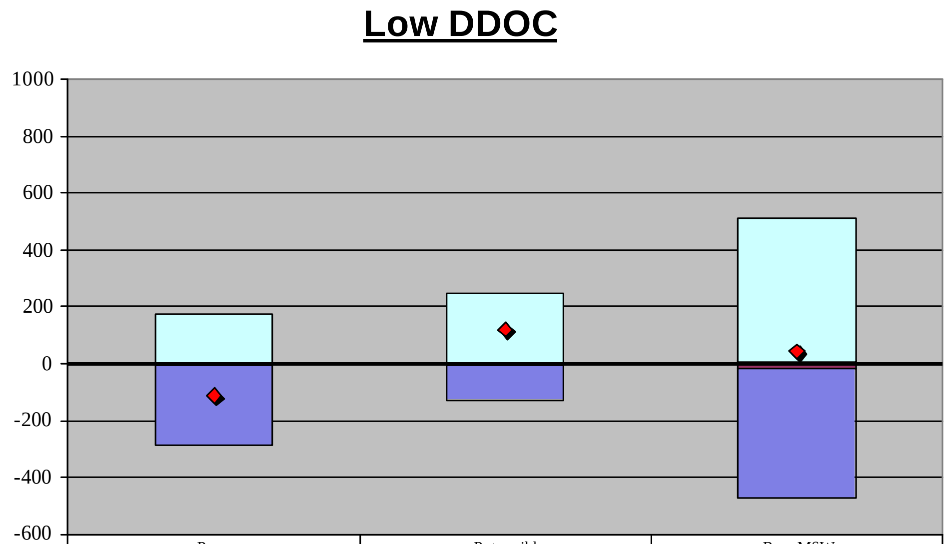
Net fluxes are shown by the diamond symbols. Putrescible wastes show the highest net flux. Paper and putrescible wastes account for 20 and 70 % respectively of net greenhouse gas flux from landfilled MSW. The net flux from paper is lower since a greater proportion of degradable carbon is sequestered in the landfill. The net flux from mixed MSW amounts to about 330 kg CO_2 eq/tonne MSW. Plastic wastes are inert in landfills and so do not contribute to methane emissions and, since all of their carbon is of fossil origin, there is no carbon sequestration attributable to this component either. Fluxes various MSW components are given in Appendix 2.

The sensitivity of the greenhouse gas fluxes to alternative views of the proportion of carbon that dissimilates in the landfills (ie DDOC) is shown by Figure 8.

Figure 8: Impact of DDOC on greenhouse gas fluxes due to landfilling untreated MSW, paper and putrescible waste (kg CO2 eq / tonne MSW).



	Paper	Putrescible	Raw MSW
□ CH4	426	402	920
■ carbon sequestered	-175	-53	-286
■ transport CO2	2	2	7
■ avoided energy and materials	-13	-12	-28
■ energy use CO2	0	0	1
◆ Net flux	241	340	614

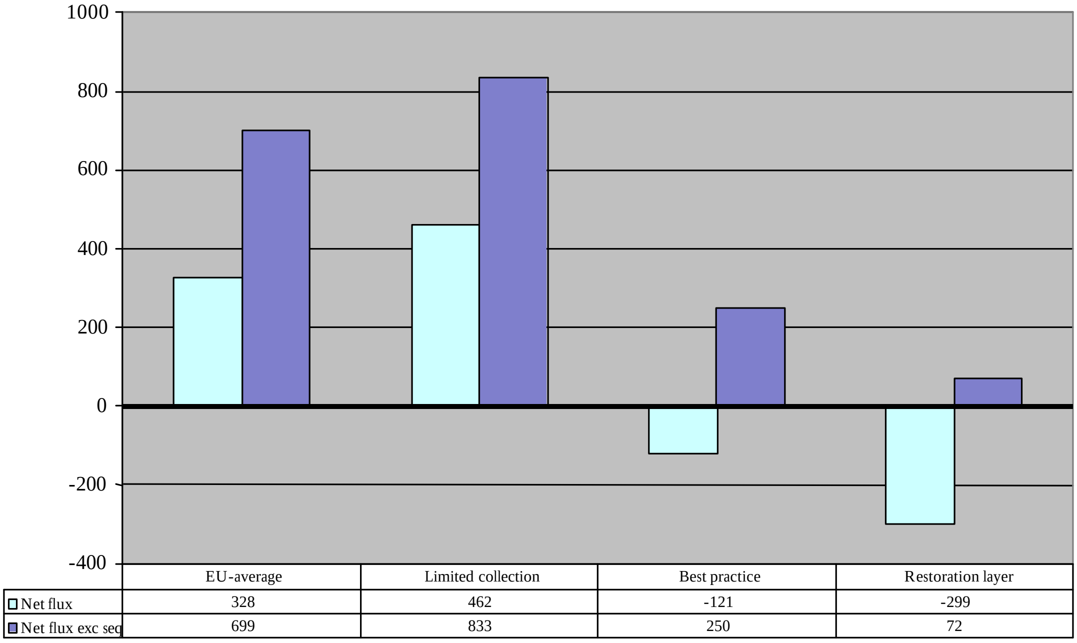


	Paper	Putrescible	Raw MSW
□ CH4	171	243	505
■ carbon sequestered	-279	-118	-456
■ transport CO2	2	2	7
■ avoided energy and materials	-5	-7	-16
■ energy use CO2	0	0	1
◆ Net flux	-112	120	42

The graphs show that for alternative assumption of the value of DDOC the total greenhouse gas flux for MSW can range from about 40 to over 650kg CO_2 eq/tonne waste.

Further uncertainty is seen regarding the impact of gas collection efficiency and the use of a ‘restoration layer’ to increase the rate of methane oxidation at the landfill surface. Figure 9 compares the total net greenhouse gas flux from untreated MSW landfilled in sites corresponding to EU-average gas collection (‘Base case’, left hand columns) with alternative sets of assumptions. The ‘limited collection’ case refers to the use of peripheral landfill gas collection intended to prevent gas migration into surrounding land, rather than to reduce surface emissions *per se*. Negative fluxes are seen for the ‘best practice’ and ‘restoration layer’ cases because the negative flux due to carbon sequstration (about –370 kg CO_2 eq/tonne, from Figure 7) is greater for these types of sites than the positive flux due to methane emissions. The graph also shows greenhouse gas fluxes where carbon sequestration is ignored (right hand columns). The net flux excluding carbon sequstration reduces with improving standards of gas management mostly through reduced emissions of methane but also through more extensive use of methane for electricity generation.

Figure 9: Total greenhouse gas fluxes from landfilling of untreated MSW, with and without credit for carbon sequestration (base case DDOC, flux in kg CO2 eq/tonne MSW).



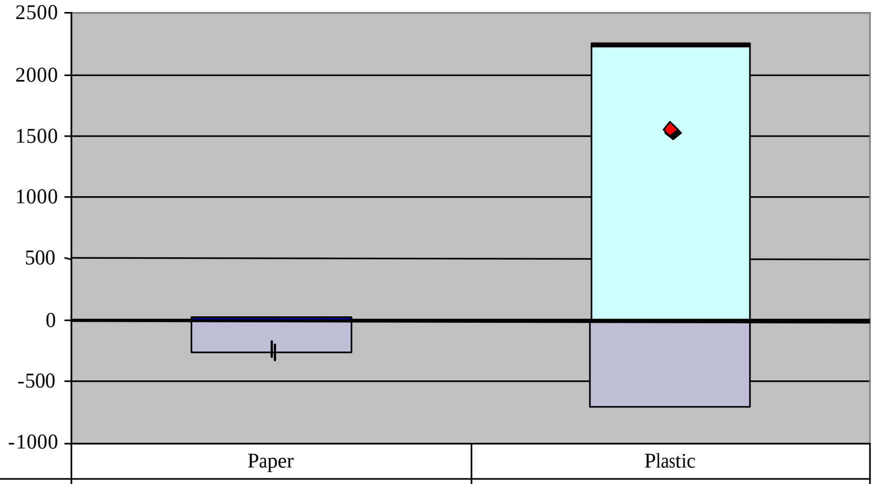
Improving the management of landfills to reduce methane emissions would clearly result in a major reduction of greenhouse gas emissions from this option, even converting a net source into a net sink for greenhouse gases, if carbon sequstration is taken into account.

3.1.2 Incineration

The breakdown of greenhouse gas fluxes from incineration depends strongly on the material being combusted, which is illustrated in the comparison between paper and plastic, incinerated in mixed MSW, shown in Figure 10. Energy is recovered as electricity only in this example. The replaced energy is assumed to come from the average EU-2000. Emission savings due to avoided energy and materials (ie the electricity that would have been generated by other means) are important for both paper and plastic waste. However, the main component is process CO_2, resulting from fossil CO_2 emissions from plastic waste. Emissions due to transportation of waste and residues and N_2O emissions from combustion together make up less than 10% of the net fluxes.

There is a net saving (negative net flux) in greenhouse gas emissions when paper is incinerated of about –235 kg CO_2 eq/tonne paper, dominated by emissions avoided from replaced energy, but in the case of plastic waste, this is more than off-set by the large process emission of fossil CO_2, resulting in a net positive flux of about 1560 kg CO_2 eq/tonne plastic.

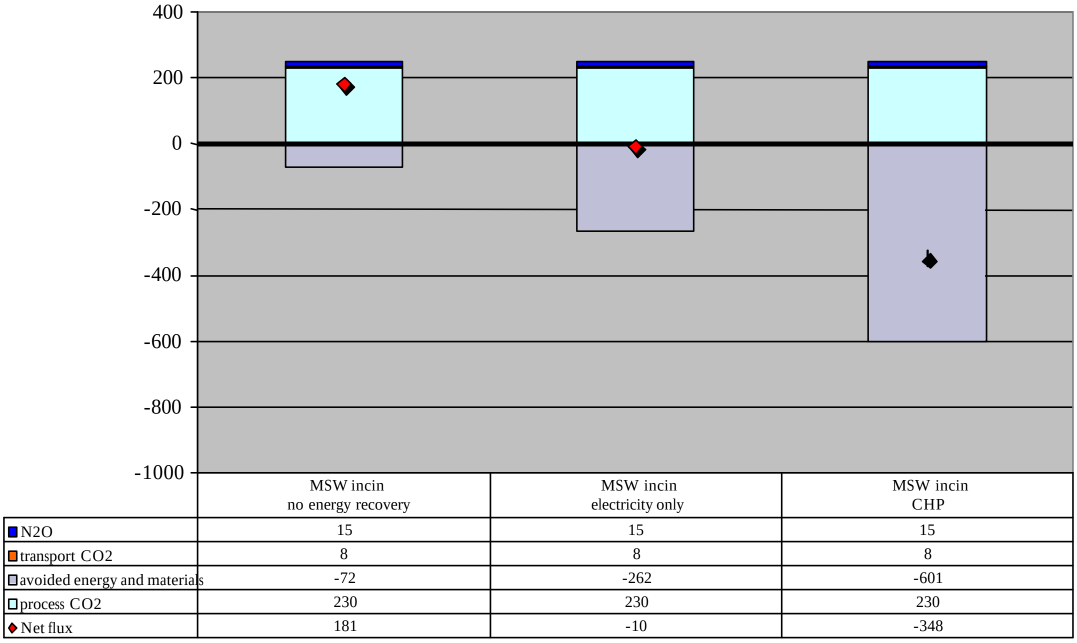
Figure 10: Greenhouse gas fluxes from mass-burn incineration of plastics and paper, replaced energy is EU-average electricity generation, kg CO2 eq/tonne of paper or plastic.



	Paper	Plastic
🟦 N2O	15	15
🟧 transport CO2	8	8
▨ avoided energy and materials	-257	-703
□ process CO2	0	2237
I Net flux	-235	1556

The greenhouse gas fluxes from incinerating bulk MSW are shown in Figure 11. The graph compares the impact of alternative assumptions on the type of energy recovery used. In the left-hand bar, where incineration takes place without energy recovery, the greenhouse gas benefits of avoided energy and materials are limited to those associated with recycling the ferrous metal recovered from the combustion residues. Overall, there is a net positive greenhouse gas flux from this option of about 180 kg CO_2 eq/tonne. The central bar represents the case where electricity only is recovered during incineration and EU-2000 average power generating mix is replaced (in addition to the benefits from ferrous metal recovery). However, the avoided emissions are almost completely neutralised by process fluxes of fossil CO_2 from plastic waste, resulting in a virtually neutral (-10 kg CO_2 eq/tonne) net greenhouse gas flux. The right-hand bar shows the situation where both heat and power are recovered from mass-burn MSW incineration. The additional recovery of heat (displacing average EU-heat sources) leads to an overall negative net greenhouse gas flux of some –350 kg CO_2 eq/tonne.

Figure 11: Greenhouse gas fluxes from mass-burn incineration of MSW. Displaced energy is EU-2000 average electricity and heat sources. (kg CO2 eq/tonne MSW).



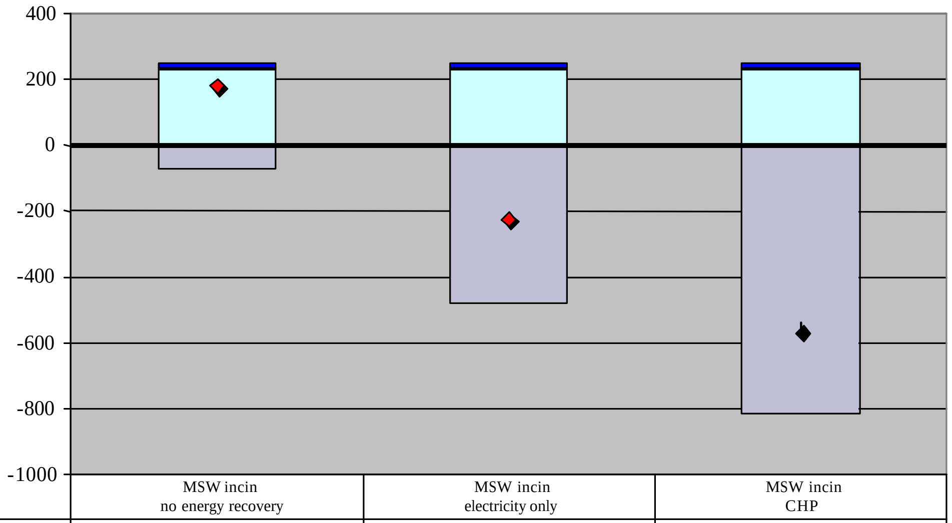
	MSW incin no energy recovery	MSW incin electricity only	MSW incin CHP
■ N2O	15	15	15
■ transport CO2	8	8	8
□ avoided energy and materials	-72	-262	-601
□ process CO2	230	230	230
◆ Net flux	181	-10	-348

The graph shows that energy recovery can make a substantial contribution to the net greenhouse gas flux from mass-burn incineration, but the magnitude of this effect depends on the energy sources that are replaced. In the above example, the avoided emissions associated with energy were assumed to come from EU-average plant mix. Two alternative views are now considered – firstly the displacement of coal-fired electricity generation and secondly, wind powered generation. In both instances the replaced heat source remains as average EU-2000 heat mix.

Figure 12 (upper section) shows the results for when coal-fired electricity is replaced. This results in a marked reduction in net greenhouse gas fluxes due to MSW incineration for electricity and CHP applications to –225 and –560 kg CO_2 eq/tonne.

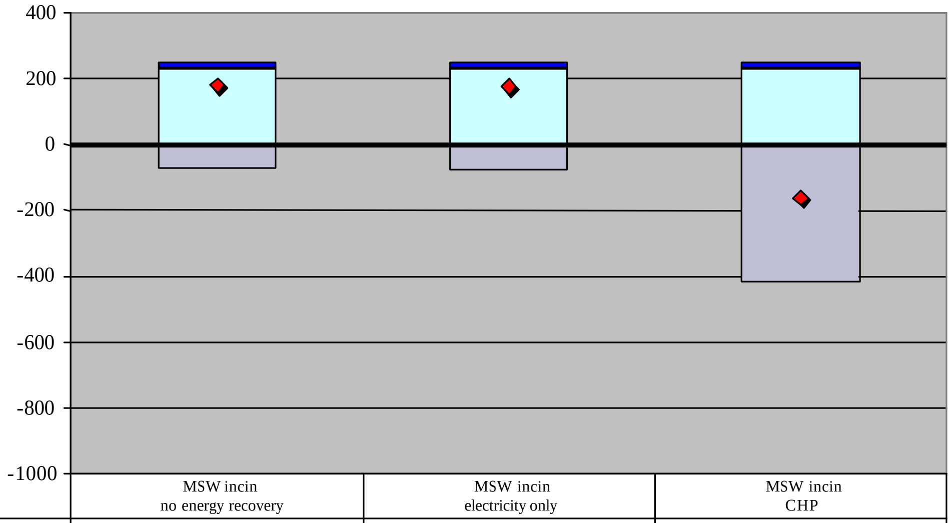
However, when wind-powered electricity generation is replaced (lower section in Figure 12), electricity-only incinerators perform almost as badly in greenhouse gas flux terms as those without any energy recovery (net greenhouse gas flux of about 180 kg CO_2 eq/tonne), although CHP plant (with heat replacing EU-average) still shows a negative net flux of about –160 kg CO_2 eq/tonne.

Figure 12: Impact of source of replaced electricity on greenhouse gas fluxes from mass-burn incineration of MSW. EU-2000 average heat sources is assumed for CHP options. (kg CO2 eq/tonne).



Electricity from coal-fired power stations replaced

	MSW incin no energy recovery	MSW incin electricity only	MSW incin CHP
■ N2O	15	15	15
■ transport CO2	8	8	8
□ avoided energy and materials	-72	-478	-816
□ process CO2	230	230	230
◆ Net flux	181	-225	-563



Electricity from wind turbines replaced

	MSW incin no energy recovery	MSW incin electricity only	MSW incin CHP
■ N2O	15	15	15
■ transport CO2	8	8	8
□ avoided energy and materials	-72	-76	-414
□ process CO2	230	230	230
◆ Net flux	181	177	-161

Pyrolysis and gasification of bulk MSW with energy recovered as electricity only produce virtually identical greenhouse gas fluxes as mass-burn incineration, and so are not described further. More information is provided in Appendix 3.

Mass-burn incineration of MSW, even without energy recovery, gives lower net greenhouse gas fluxes than landfilling with EU-average standards of gas collection. With best practice standards of landfill site management, incineration with CHP energy recovery has the lower net flux (assuming that average EU-2000 electricity and heat sources are replaced, and taking account of carbon sequestered in the landfills). If carbon sequestration in landfills is not included, then MSW incineration with energy recovered as electricity only, as well as CHP plants, have lower fluxes than the best standards of landfills. The advantage of incineration depends crucially on the sources of energy it replaces. If coal-fired electricity generators are replaced, then electricity-only incinerators have lower greenhouse gas fluxes than even the best types of landfills, and even more so in the case of CHP incinerators. However, if wind turbines are the replaced generators, then the best standards of landfills have a lower net flux than CHP incinerators.

3.1.3 Refuse Derived Fuel (RDF) combustion

An alternative to mass-burn incineration that has found limited application is in the production and combustion of RDF, made from paper, plastics and textiles. In this analysis, we consider the greenhouse gas fluxes of RDF burnt in either a dedicated combustor (in this instance, we assume a fluidised bed combustor (FBC), probably co-firing with wood, peat or coal) or as a substitute for coal in a cement kiln or pulverised fuel powers station. Here we assume that the RDF displaces coal on an equivalent energy content basis and that there is no change in thermal efficiency associated with co-firing. The issue of alternative choices for replaced generation emissions is therefore not relevant in these latter cases. However, there is some uncertainty over what would be replaced in the FBC instance. Clearly for a pre-existing plant, if RDF were not used then more wood, peat or coal would be required. However, if a new facility were to be built, it could substitute for new renewable electricity technology with a much lower emission factor, such as wind or biomass, so reducing the net greenhouse gas benefit of RDF combustion. In this instance, we have assumed that the RDF-FBC electricity replaces power that would otherwise have come from EU-2000 average plant mix.

This analysis also considers fluxes from disposal of the residual MSW to landfill and those associated with recovering metals for recycling during the RDF preparation.

Greenhouse gas fluxes from RDF combustion are shown in the three left-hand bars in Figure 13. For RDF combustion in FBC boilers, there is a net positive greenhouse gas flux of about 70 kg CO_2 eq/tonne, with the energy benefit being outweighed by the process emission of fossil CO_2 from plastic combustion. Where RDF replaces coal in pulverised fuel power stations or in cement kilns, the displaced energy benefits are much larger and as a result a net negative greenhouse gas flux of about –340 kg CO_2 eq/tonne of MSW occur.

The RDF options also incur greenhouse gas fluxes from the disposal of residual MSW (mostly putrescible waste, but also glass and miscellaneous materials) and from the recycling of metals recovered during sorting and fuel preparation. We consider the issue of residual MSW components first.

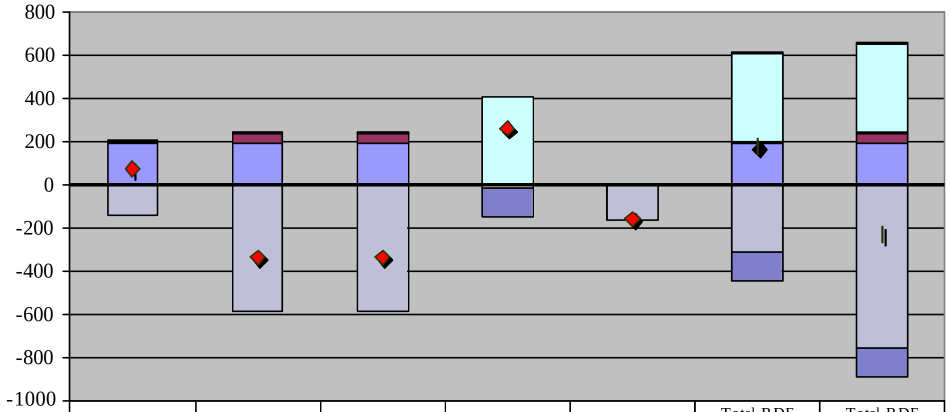
Net greenhouse gas emissions from the landfill of MSW residues are dominated by methane emissions, which are only partly offset by carbon sequestration, resulting in a positive net greenhouse gas flux of about 260 kg CO_2 eq/tonne MSW. Base case levels of DDOC and EU-average gas collection are assumed for the landfill of MSW residue.

Recovery and recycling of metals (assumed to be 4% ferrous and 1% aluminium in MSW) from the MSW processed to RDF provides a significant saving in emissions of about –160 kg CO_2 /tonne MSW.

Finally, the overall greenhouse gas fluxes from the RDF combustion, plus landfilling of residual MSW and metals recovered for recycling are shown in the two right-hand bars. RDF combustion in cement kilns and power stations produce the same outcome and so only one set of data are shown for these situations. Overall, RDF combustion in the FBC system gives rise to an overall positive greenhouse gas flux (about 180 CO_2 eq/tonne MSW), compared with a net negative flux of about –230 kg CO_2 eq/tonne for co-firing as a coal substitute in cement kilns or pulverised fuel power stations.

Figure 13: Greenhouse gas fluxes from refuse-derived fuel (RDF) combustion and landfill disposal of residual MSW, kg CO2 eq/tonne MSW.

The results are based on EU-average electricity as the replaced energy source, except for combustion in power stations or cement kilns, where coal is assumed to be replaced. Landfill fluxes are calculated for base case DDOC and EU-average gas control.



	RDF-FBC	RDF-power station	RDF-cement	RDF residue landfill	Metal recycling	Total RDF-FBC/landfill	Total RDF-cement/landfill
■ N2O	12	6	6	0	0	12	6
□ CH4	0	0	0	404	0	404	404
■ carbon sequestered	0	0	0	-134	0	-134	-134
□ transport CO2	3	3	3	4	1	8	8
■ avoided energy and materials	-138	-585	-585	-12	-160	-310	-757
■ energy use CO2	0	43	43	1	1	1	45
■ process CO2	196	196	196	0	0	196	196
◆ Net flux	73	-337	-337	262	-158	177	-233

RDF combustion currently accounts for less than 10% of waste incinerated. Where the RDF is combusted as a direct replacement for coal in power stations or cement kilns, and the residue is disposed of to landfill operated to EU-average standards, then an overall negative greenhouse gas flux results. The greenhouse gas flux is much smaller than that resulting from the disposal of bulk MSW to EU-average standard landfills, but of similar magnitude to that achievable through the highest standards of landfill gas management. Where the RDF replaces EU-average electricity sources, then the greenhouse gas flux from the landfilled residue outweighs the negative flux from the incineration step and RDF results in an overall positive net greenhouse gas flux. This is only slightly less than that achieved by landfilling the bulk MSW, with EU-average landfill gas management.

3.1.4 Mechanical Biological Treatment (MBT)

Two main options for MBT have been analysed – MBT treatment followed by landfilling of the compost and diversion of a proportion of the paper, plastics, textiles, miscellaneous combustible and miscellaneous non-combustibles to (a) either direct landfilling or (b) mass-burn incineration with energy recovered as electricity. The proportion of rejects diverted was as follows:

• Paper	20%
• Plastics	100%
• Textiles	50%
• Miscellaneous combustibles	50%
• Miscellaneous non-combustibles	100%

Full details of the MBT process assumptions are given in Appendix 4. Materials sent to landfill directly were assumed to decay with base case DDOC and with EU-average gas collection efficiency. Electricity generated at the incinerator was assumed to displace EU-2000 average generation mix. In both cases, all of the metals (assumed to be 4% ferrous and 1% aluminium in MSW) were recovered for recycling.

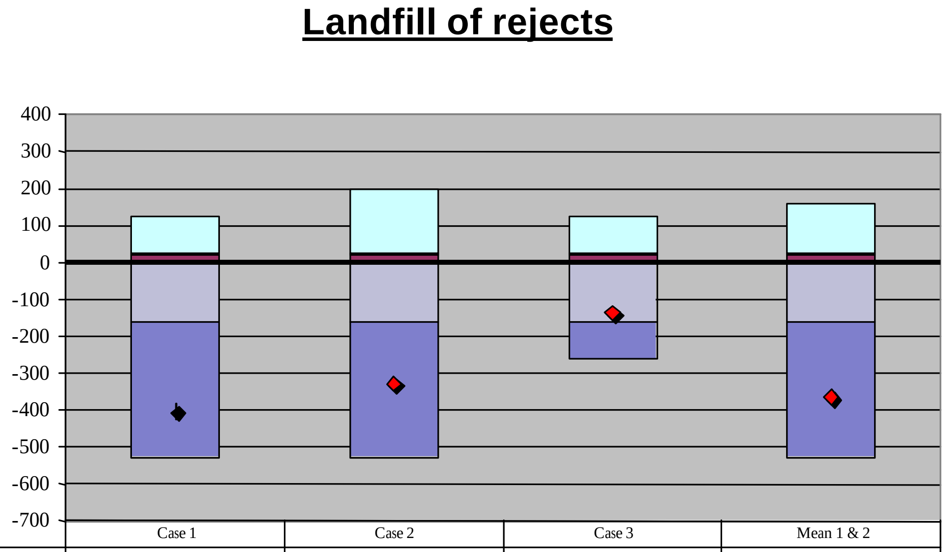
The behaviour of the compost following MBT is examined under three sets of assumptions:

- **Case 1.** The highly stabilised compost is landfilled and all of the small amount of methane formed is oxidised before it escapes.
- **Case 2.** Less completely stabilised compost is landfilled and 25% of the methane formed escapes, the rest is oxidised.
- **Case 3.** The highly stabilised compost is used as a surface dressing for landfill site restoration and decays aerobically. Because further decomposition occurs under the aerobic conditions under which the compost is used, sequestration of carbon is limited to the much lower rate used for compost applied to agricultural land.

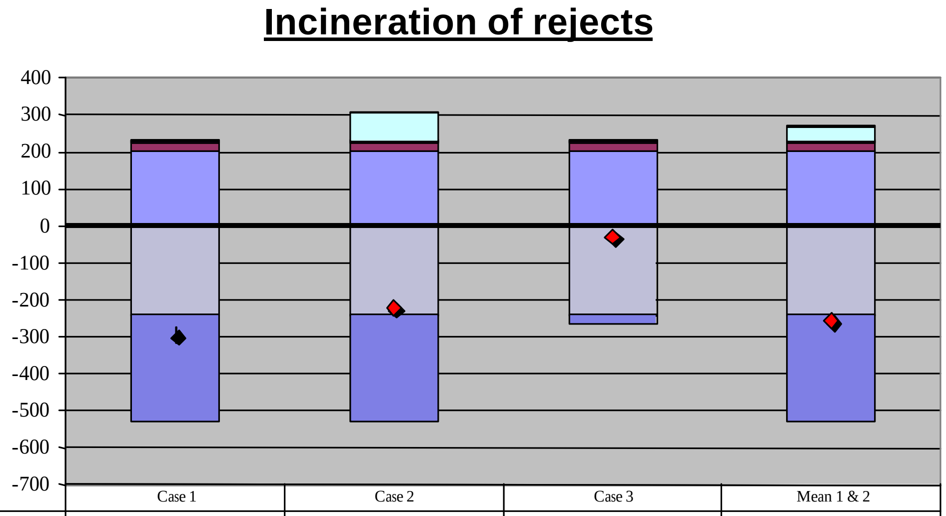
Cases 1 and 2 are thought to represent the most likely end uses for MBT compost once the technique becomes established. Case 3 is most likely to be important when MBT is initially established on a working landfill, but opportunities for disposing of compost as surface dressing will diminish as the option becomes more firmly established. Results are presented for all three cases separately, and as an average of cases 1 and 2, since for the reasons just given, these are thought to most closely represent the major fate of the compost. Levels of methane production from MBT compost were assumed to be too low to allow energy recovery.

We will consider first the case of MBT with landfill of rejects. The greenhouse gas fluxes associated with this option are shown in the upper graph of Figure 14. Net greenhouse gas emissions are strongly negative for cases 1 and 2, which benefit from the large amounts of carbon sequestered, and less so for case 3, with lower levels of sequestration. The negative fluxes associated with avoided energy and materials come from the recycling of metals recovered from the process. The methane emissions in Case 1 and Case 3 originate almost entirely from the 20% of paper that is removed for direct landfilling before the composting stage, although emissions from MBT compost also contribute in Case 2. Direct landfilling of paper also contributes to the relatively large component of carbon sequestration seen in Case 3, where the compost itself is used for site restoration and so there is relatively little carbon sequestration associated with the compost in this application.

Figure 14: MBT with landfill or incineration of rejects and recycling of metals (kgCO2 eq/tonne MSW).


Landfill of rejects

	Case 1	Case 2	Case 3	Mean 1 & 2
■ N2O	0	0	0	0
■ CH4	97	171	97	134
■ carbon sequestered	-364	-364	-99	-364
■ transport CO2	5	5	5	5
■ avoided energy and materials	-162	-162	-162	-162
■ energy use CO2	22	22	22	22
■ process CO2	0	0	0	0
◆ Net flux	-403	-329	-137	-366


Incineration of rejects

	Case 1	Case 2	Case 3	Mean 1 & 2
■ N2O	3	3	3	3
■ CH4	0	74	0	37
■ carbon sequestered	-289	-289	-23	-289
■ transport CO2	5	5	5	5
■ avoided energy and materials	-241	-241	-241	-241
■ energy use CO2	22	22	22	22
■ process CO2	205	205	205	205
◆ Net flux	-295	-221	-30	-258

Turning now to the option of MBT with incineration of rejects, shown in the lower graph of Figure 14, the results show a slighly smaller net negative flux for the three cases considered, compared with the MBT with landfill of rejects discussed above. The benefits of avoided materials and energy which are dominated by the avoided fluxes associated with metal recycling are in this case supplemented by the relatively small contribution of avoided emissions due to electricity generation at the incinerator. However, this is outweighed by the increase in process emissions due to fossil CO_2 emissions from plastics incineration, so that the overall net (negative) flux is slightly less than the corresponding results for MBT with landfill of rejects. In this option, all of the methane emissions and carbon sequestration are attributable to the compost iteslf, since all rejects are incinerated,rather than going to landfill.

The main advantage of MBT over landfilling of untreated MSW lies in the much lower amount of methane emitted from the stabilised waste, along with avoided emissions due to recycling metals and energy recovery from incineration of rejects. On the other hand, incineration of plastic rejects contributes to process-derived emissions. Overall, MBT results in a lower greenhouse gas flux than EU-average standard of landfilling when the composted residue is landfilled to maximise carbon sequestration, but is comparable with fluxes from the highest standards of landfill. The relative performance of specific MBT schemes will depend on flows of various waste streams and so these examples provide only a guide. Non-greenhouse gas benefits of MBT over landfilling of raw waste include more effective use of landfill void space, greater stabilisation of the waste, reducing the threat from leachate escape and the consequent reduction in environmental management costs at the landfill. Benefits also accrue from the recycling of materials recovered in the option. MBT is less sensitive than incineration to assumptions about the source of replaced energy.

3.2 OPTIONS FOR SOURCE-SEGREGATED WASTES

Having considered the main waste management options for bulk MSW or MSW subject to only minimal segregation following collection, we will now examine greenhouse fluxes from waste segregated at source. The analysis is based on the assumption that the segregated wastes are completely removed from the MSW stream for recycling or composting / AD (in the case of putrescible wastes), and that only residual material (consisting of fines - ie dust - and miscellaneous combustible and non-combustible materials) remain. This residue is then disposed of to landfill without further treatment.

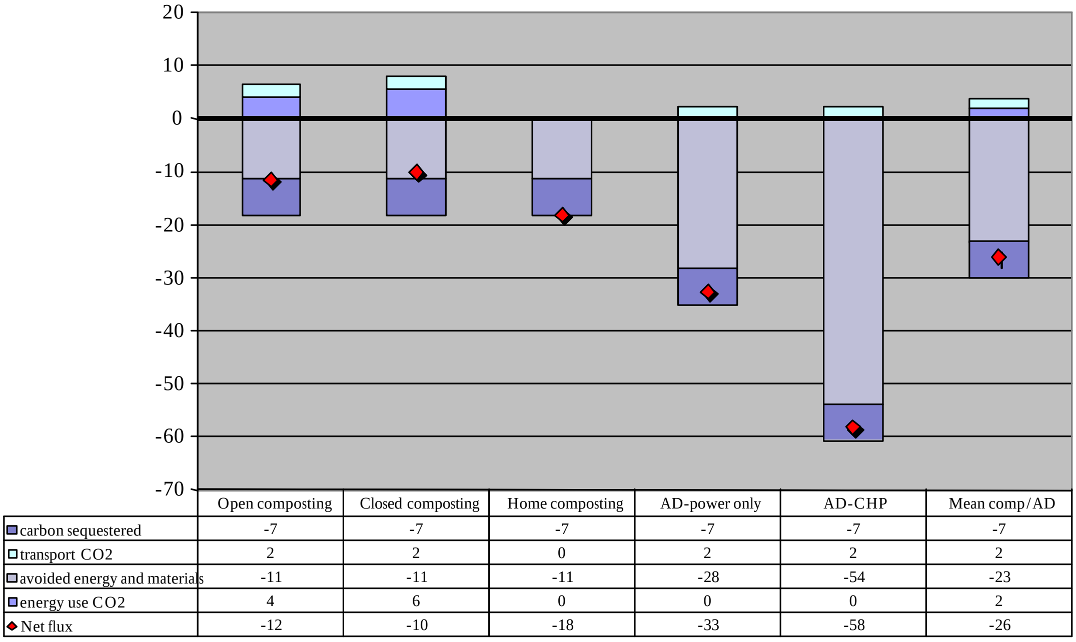
3.2.1 Putrescible wastes

Figure 15 shows the greenhouse gas fluxes associated with composting and AD of putrescible wastes – ie food and garden wastes. The main component of the net negative flux for each option is from avoided energy and materials use – which comprise benefits due to the reduced need for peat, mineral fertilisers and, in the case of AD, energy also. A relatively small additional benefit comes from carbon sequestration due to the storage of carbon in soil following the use of compost in agriculture. Organic matter in aerobic soils turns over much more rapidly than under anaerobic conditions in landfills, which accounts for the lower sequestration benefit. Details of assumptions for composting and AD are given in Appendices 5 and 6.

Overall, the AD options have the greatest net greenhouse gas flux, especially in the CHP version. The results are presented for the case where the electricity replaced by AD comes from average EU generating plant mix. Smaller benefits would be seen if wind energy were the displaced generation source and in this case the greenhouse gas benefits of AD would be similar to those of composting. Correspondingly larger benefits would accrue if coal-fired power generation were displaced. Positive greenhouse gas fluxes come from transportation and, in the case of open and closed costing systems, energy used in the process. In the case of AD, energy use is included in the overall energy efficiency of the process and so is not shown separately. For home composting there are no transport or energy use emissions.

Overall, the net greenhouse gas fluxes associated with the management of separated putrescible waste range from –12 to –58 kg CO_2 eq/tonne MSW, with an overall average of –26 kg CO_2 /tonne MSW, shown in the right-hand column of the graph.

Figure 15: Greenhouse gas fluxes from composting and AD of putrescible wastes, assuming average EU electricity replaced. kg CO2 eq/tonne MSW.

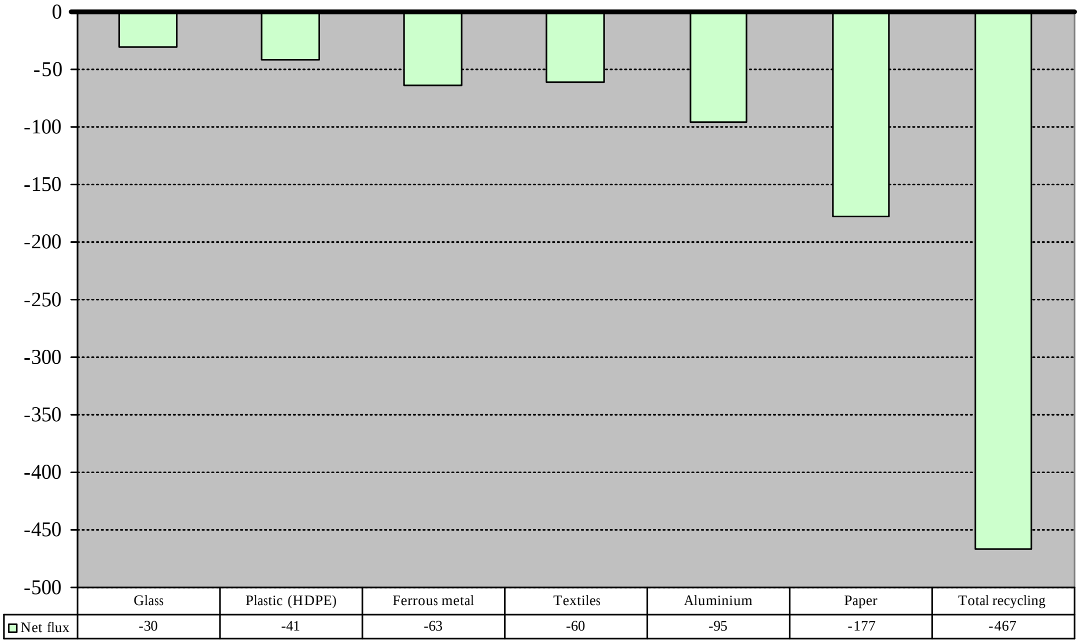


	Open composting	Closed composting	Home composting	AD-power only	AD-CHP	Mean comp/AD
▨ carbon sequestered	-7	-7	-7	-7	-7	-7
☐ transport CO2	2	2	0	2	2	2
▨ avoided energy and materials	-11	-11	-11	-28	-54	-23
▨ energy use CO2	4	6	0	0	0	2
◆ Net flux	-12	-10	-18	-33	-58	-26

3.2.2 Recycling

Greenhouse gas fluxes from recycling source-segregated materials are summarised in Figure 16, based on background information given in Appendix 7. Appendix 7 also shows the greenhouse gas fluxes expressed per tonne of each material recovered. Here we show the flux associated with recycling each component per tonne of MSW. The results for plastics are based on HDPE. Paper and aluminium offer the 'best' returns from recycling per tonne of MSW, compared with glass, which offers the least benefit. The total flux from recycling is shown in the right-most bar, which is the sum of the individual waste components. Recycling all of the materials indicated would result in a net negative greenhouse gas flux (ie a saving) of about –470 kg CO_2 eq/tonne MSW.

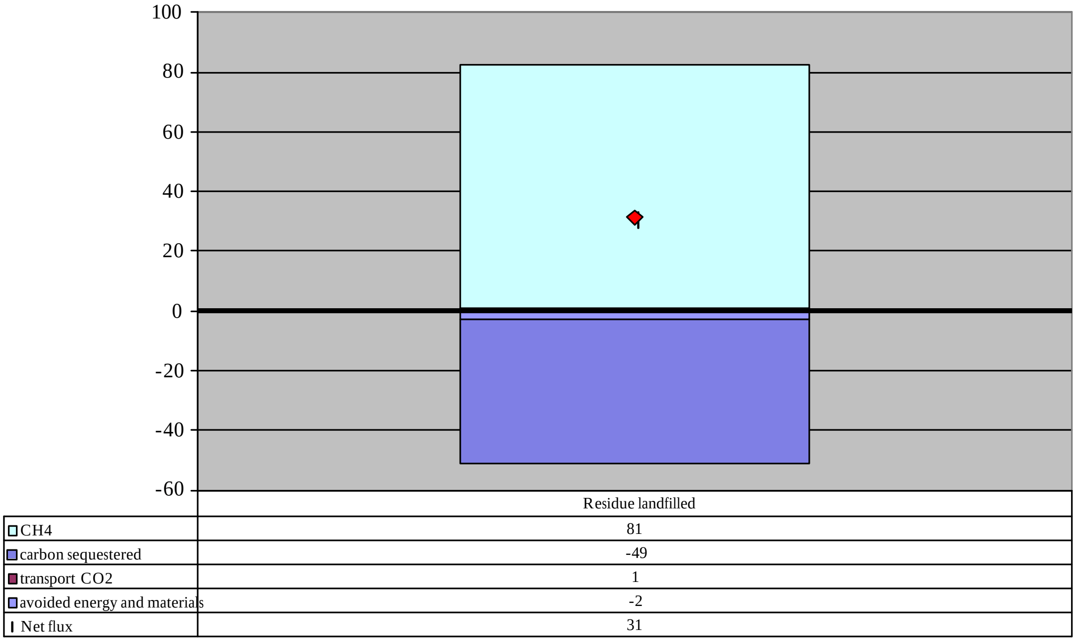
Figure 16: Greenhouse gas fluxes from recycling, kg CO2 eq/tonne MSW.



	Glass	Plastic (HDPE)	Ferrous metal	Textiles	Aluminium	Paper	Total recycling
🟩 Net flux	-30	-41	-63	-60	-95	-177	-467

In addition to the fluxes associated with recycling source-segregated materials, we also need to take account of the fluxes associated with disposal of the residual material, in this case to landfill. These results are shown in Figure 17, based on the assumption that the landfill gas collection is managed to EU-average standards. A net flux of about 30 kg CO_2 eq/tonne MSW results from this stage in the process, if we count the benefit due to carbon sequestration, or 80 kg CO_2 eq/tonne MSW without.

Figure 17: Greenhouse gas flux from recycling residue disposed of to landfill kg CO2 eq/tonne MSW.



	Residue landfilled
☐ CH4	81
■ carbon sequestered	-49
■ transport CO2	1
■ avoided energy and materials	-2
I Net flux	31

From these results, we can now calculate the overall greenhouse gas flux associated with maximum recycling and composting for separately collected wastes and landfilling of the residue, under the base-case assumptions outlined above. This is shown in Table 10, with and without the negative flux credited to carbon sequestration (both in the landfilled residue and compost applied to soil):

Table 10: Net greenhouse gas fluxes from recycling and composting of source-separated wastes, kg CO2 eq/tonne MSW.

	Including C sequestration	Excluding C sequestration
Compost /AD	-26	-19
Recycling	-467	-467
Landfill of residue	31	80
Total flux	-461	-406

Recycling and composting / AD of source segregated wastes offer the lowest greenhouse gas fluxes of the waste management options analysed, under base case assumptions on landfill gas management and the source of replaced energy. The performance is relatively insensitive to assumptions concerning the inclusion of carbon sequestration, replaced energy and landfill site management.

3.2.3 Overall results

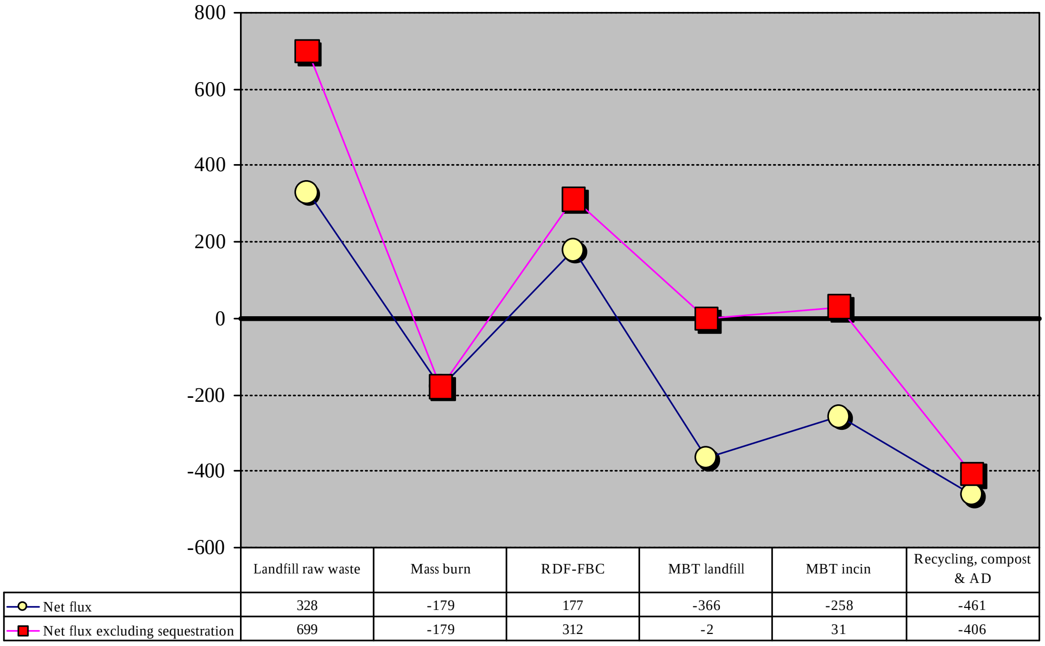
In the following graphs, the net average greenhouse gas fluxes from source separated materials managed through maximum recycling and composting / AD is compared with options for treating bulk MSW not subjected to source segregation.

Figure 18 compares the waste management options under base-case assumptions for landfill gas management and replaced electricity source. All options appear 'better'[a] than landfill of untreated MSW, with the maximum recycling option performing best of all. The results are presented with and without the credit due to carbon sequestration. Sequestration makes only a minor difference to the recycling and composting option, but has a large impact on landfill itself and the MBT options. Landfill remains the 'worst' option even when sequestration is taken into account. If sequestration is not included, then the next best option to recycling and composting is mass-burn incineration, illustrated here by the average fluxes from electricity-only and CHP applications. However, if sequestration is counted, then MBT becomes the next preferred option (with landfill of residues performing better than incineration of residues). RDF combustion is the second worst option after landfilling untreated wastes – assuming average EU electricity plant is the avoided source of electricity. The RDF options in which it replaces coal in either a pulverised fuel power station or cement kiln are not shown in these graphs. These options achieve high net negative fluxes (see Figure 13) but opportunities for their widespread deployment are very limited.

---

[a] In this context, 'better', 'best', 'worse' and 'worst' etc are used to denote the relative performance of options in terms of greenhouse gas fluxes only.

Figure 18: Overall net greenhouse gas fluxes from waste management options - EU-average landfill gas collection and EU-2000 average electricity replaced kg CO2 eq/tonne MSW.

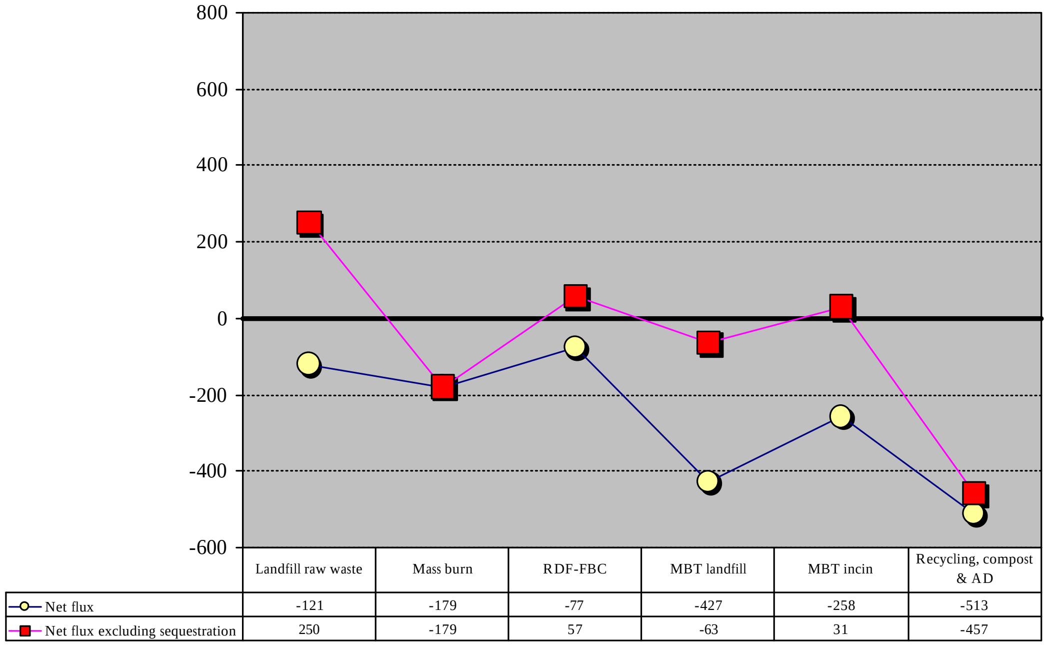


	Landfill raw waste	Mass burn	RDF-FBC	MBT landfill	MBT incin	Recycling, compost & AD
Net flux	328	-179	177	-366	-258	-461
Net flux excluding sequestration	699	-179	312	-2	31	-406

The impact of improving landfill gas collection and management through adopting ‘best practice’ and the use of ‘restoration layers’ at landfills to reduce methane escape and increase its collection for energy recovery is shown in Figure 19 and Figure 20. The maximum recycling and composting option remains the best option for both sets of assumptions, irrespective of whether or not carbon sequestration is included. With landfilling with a restoration layer, the net carbon flux (including sequestration) is lower than that of mass-burn incineration, which becomes the worst option. In this case, MBT with landfill of rejects is the next best option to recycling and composting.

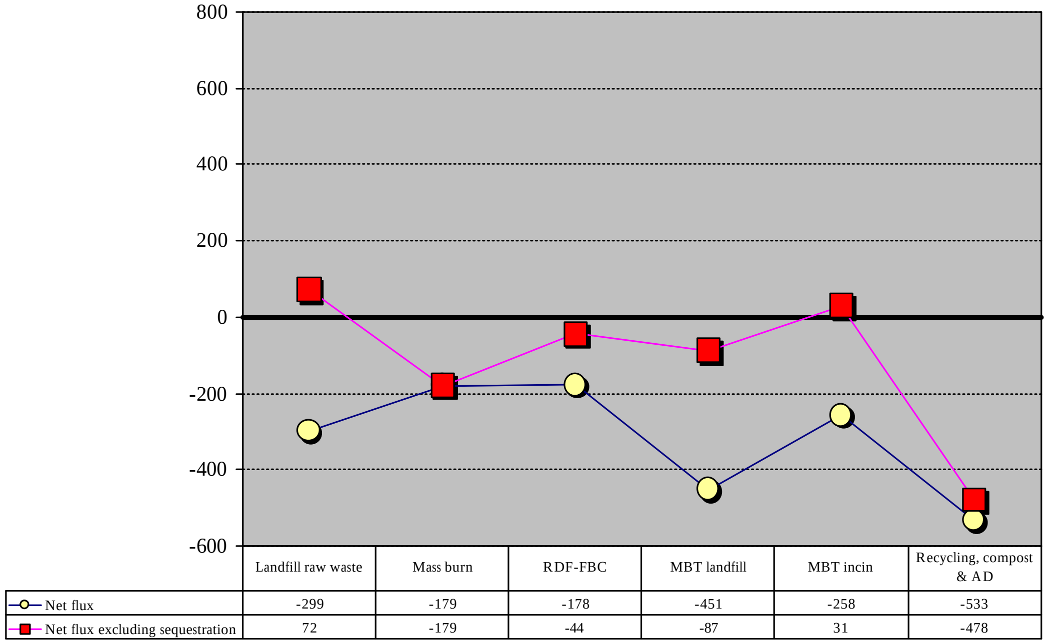
Where sequstration is not taken into account, mass-burn incineration performs second best to recycling and composting, and better than MBT or RDF combustion.

Figure 19: Overall net greenhouse gas fluxes from waste management options – best practice landfill gas collection and EU-average electricity replaced kg CO2 eq/tonne MSW.



	Landfill raw waste	Mass burn	RDF-FBC	MBT landfill	MBT incin	Recycling, compost & AD
○ Net flux	-121	-179	-77	-427	-258	-513
■ Net flux excluding sequestration	250	-179	57	-63	31	-457

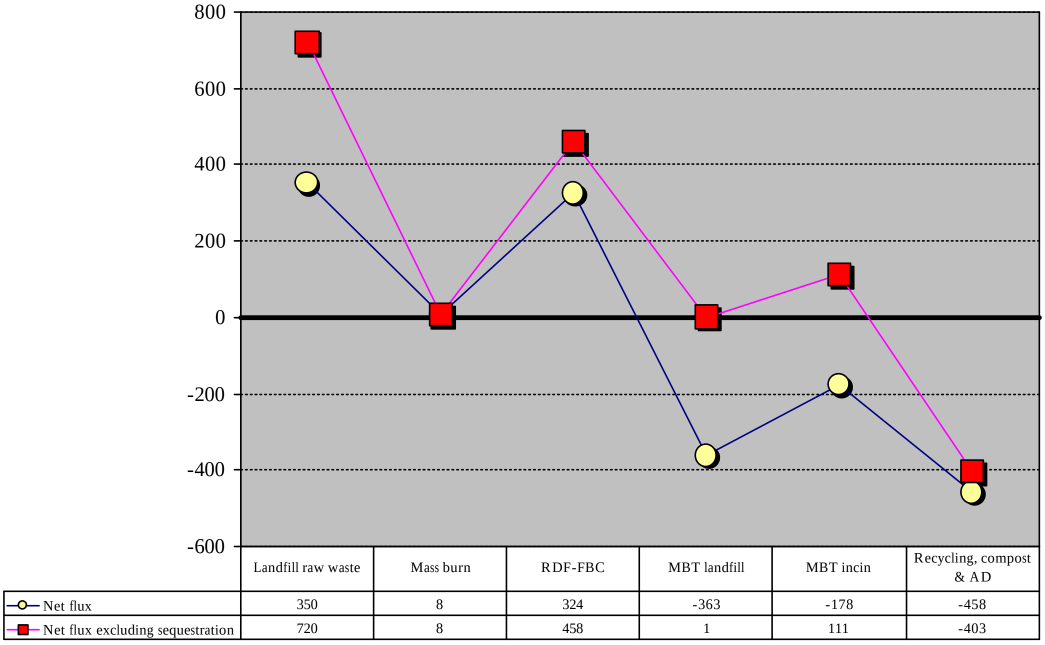
Figure 20: Overall net greenhouse gas fluxes from waste management options – 'restoration layer landfill gas collection and EU-average electricity replaced kg CO2 eq/tonne MSW.



	Landfill raw waste	Mass burn	RDF-FBC	MBT landfill	MBT incin	Recycling, compost & AD
○ Net flux	-299	-179	-178	-451	-258	-533
■ Net flux excluding sequestration	72	-179	-44	-87	31	-478

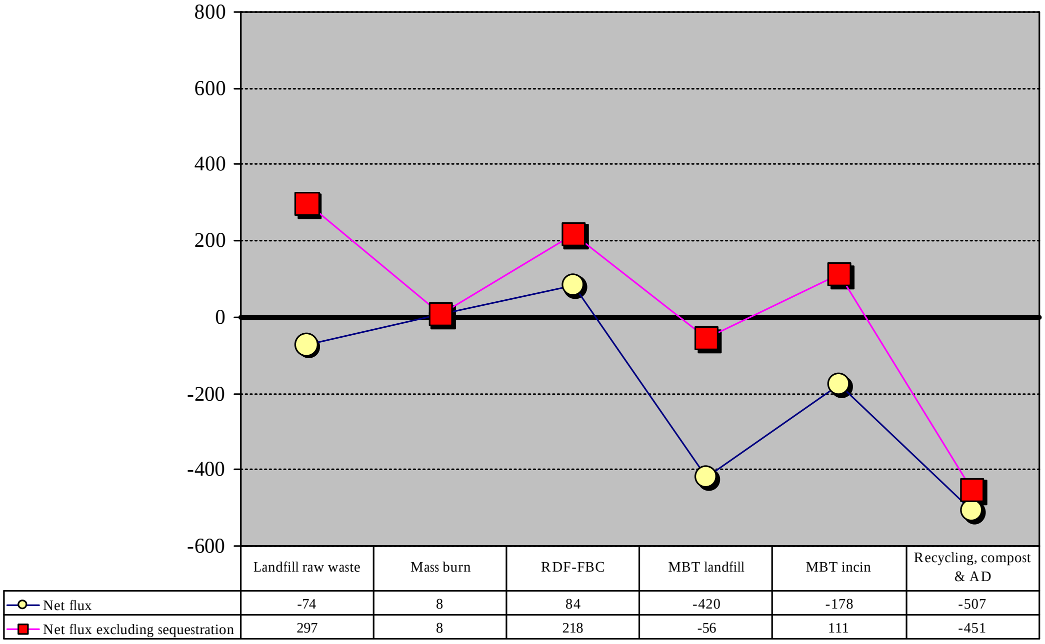
Where wind-generated electricity is the displaced energy source (see Figure 21, Figure 22 and Figure 23), recycling and composting remain the best options, irrespective of whether carbon sequstration is included or the assumed landfill gas collection efficiency.

Figure 21: Overall net greenhouse gas fluxes from waste management options – EU-average landfill gas collection and wind electricity replaced kg CO2 eq/tonne MSW.



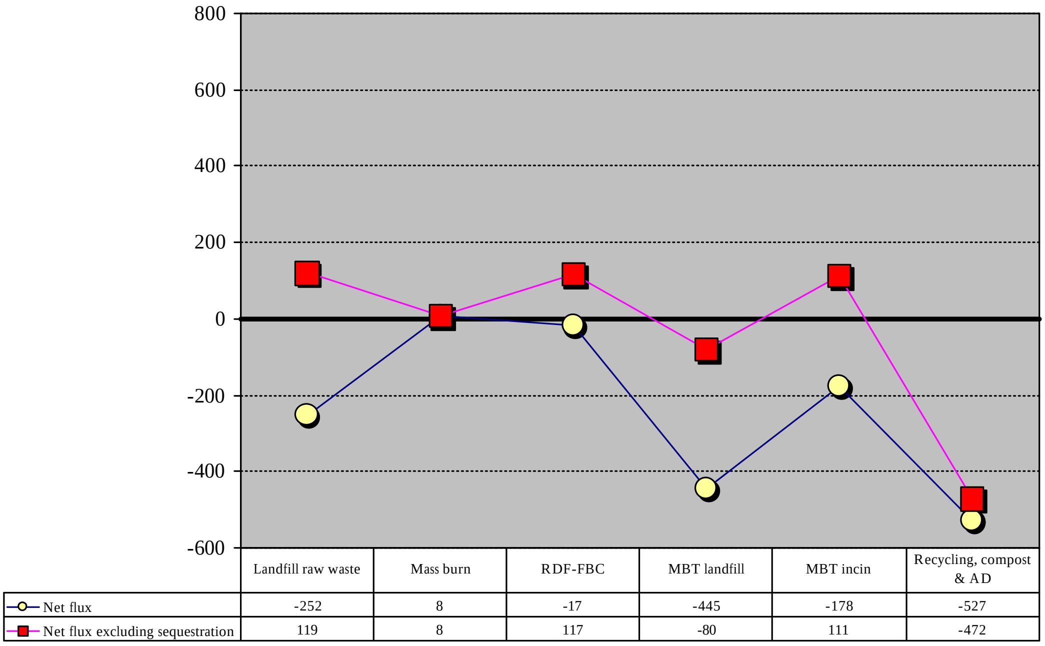
	Landfill raw waste	Mass burn	RDF-FBC	MBT landfill	MBT incin	Recycling, compost & AD
○ Net flux	350	8	324	-363	-178	-458
■ Net flux excluding sequestration	720	8	458	1	111	-403

Figure 22: Overall net greenhouse gas fluxes from waste management options – best practice landfill gas collection and wind electricity replaced kg CO2 eq/tonne MSW.



	Landfill raw waste	Mass burn	RDF-FBC	MBT landfill	MBT incin	Recycling, compost & AD
○ Net flux	-74	8	84	-420	-178	-507
■ Net flux excluding sequestration	297	8	218	-56	111	-451

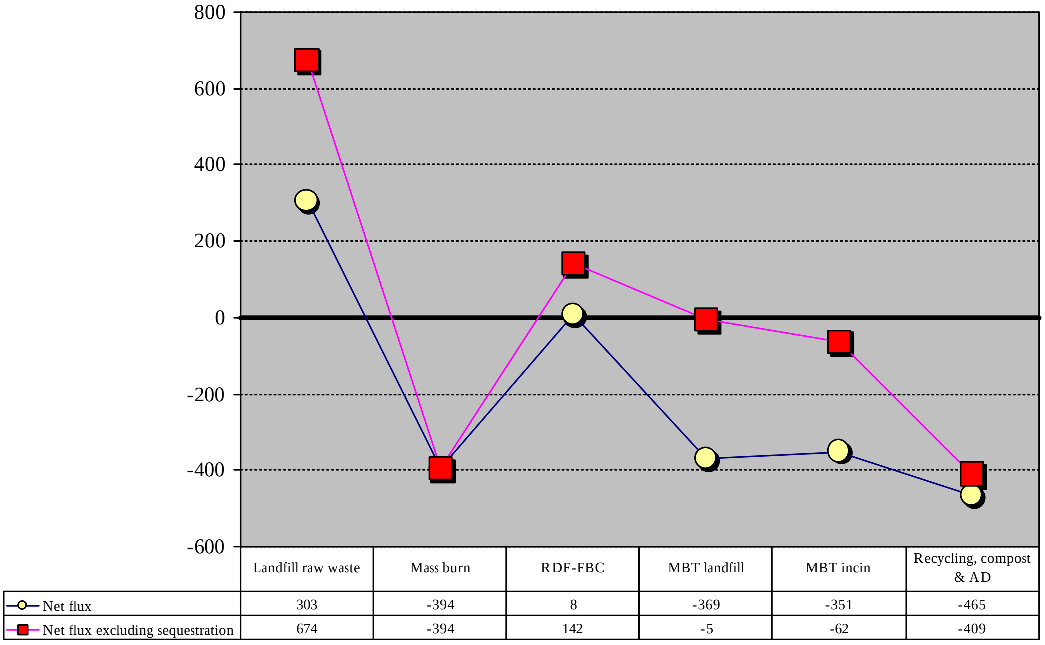
Figure 23: Overall net greenhouse gas fluxes from waste management options – restoration layer landfill gas collection and wind electricity replaced kg CO2 eq/tonne MSW.



	Landfill raw waste	Mass burn	RDF-FBC	MBT landfill	MBT incin	Recycling, compost & AD
○ Net flux	-252	8	-17	-445	-178	-527
🟥 Net flux excluding sequestration	119	8	117	-80	111	-472

In the case where coal-fired electricity is replaced, recycling and composting still provide the best overall outcome, but mass-burn incineration is the next best option, ahead of MBT and RDF (Figure 24). However, the performance of incineration is based on equal deployment of power-only and CHP schemes. In this comparison, CHP plant would give a better performance than recycling and composting – see Figure 12.

Figure 24: Overall net greenhouse gas fluxes from waste management options – EU-average landfill gas collection and coal-fired electricity replaced kg CO2 eq/tonne MSW.



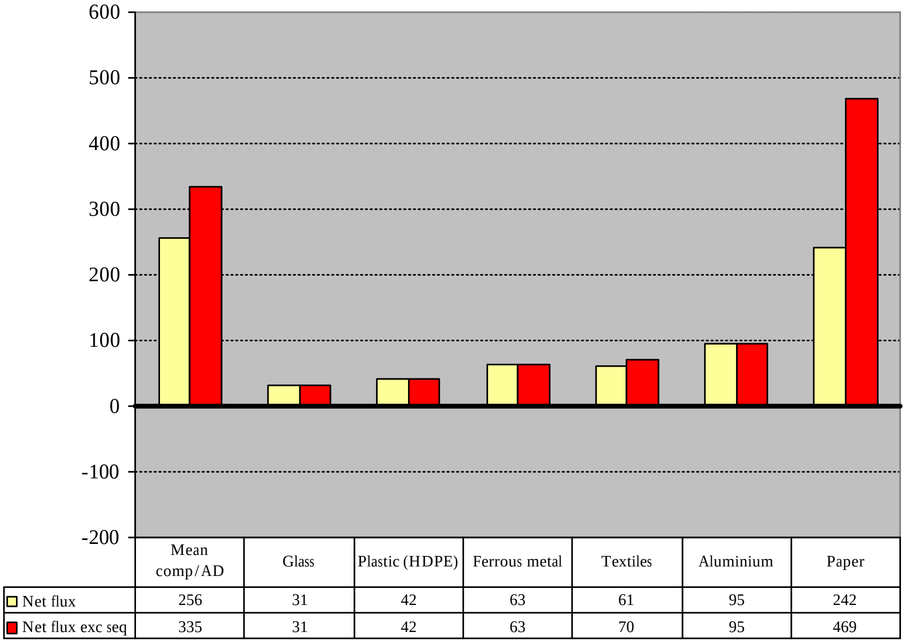
	Landfill raw waste	Mass burn	RDF-FBC	MBT landfill	MBT incin	Recycling, compost & AD
○ Net flux	303	-394	8	-369	-351	-465
🟥 Net flux excluding sequestration	674	-394	142	-5	-62	-409

Diversion from landfill to recycling

From the greenhouse gas fluxes reported above for recycling various separately-collected MSW components (including putrescible wastes diverted to composting and AD), we can now calculate the impact on overall greenhouse gas flux by diverting material to recycling that would otherwise have been landfilled without further treatment. In this instance, the overall change in flux is the difference between the net flux from landfilling the waste and the net flux from recycling it. The results are expressed on a per tonne of MSW basis, to show which waste streams, on recycling, yield the greatest improvement in greenhouse gas fluxes.

We start first with the case where waste is diverted from landfills operated to EU-average standards of gas management (Figure 25). Results are shown with and without the negative flux attributed to carbon sequestration. The greatest impact of including the sequestration terms falls on the avoided flux from landfill, since the sequestration term for composted putrescible waste is relatively small. Taking the case where sequestration is included in the net flux, the results show that diversion of putrescible wastes from landfill to composting/AD (we assume here the average treatment flux from composting or AD treatment shown in Figure 15) gives the overall greatest reduction, closely followed by paper. If carbon sequestration is excluded, greater overall savings accrue because of the worse performance of landfill, and in this case, diversion of paper, rather than putrescibles, from landfill offers the greatest benefit. Considerably smaller savings result from diverting the remaining wastes shown in the figure.

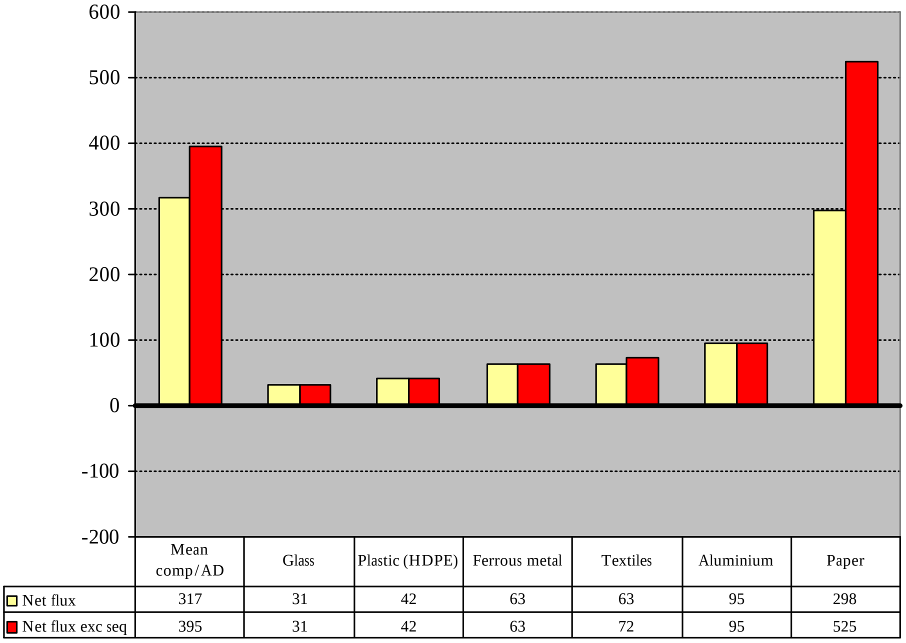
Figure 25: Overall impact of diverting waste components from landfill with EU-average standard of gas collection to recycling (or composting / AD for putrescible waste), kg CO2 eq/tonne MSW. Positive values indicate a net saving in flux.



	Mean comp/AD	Glass	Plastic (HDPE)	Ferrous metal	Textiles	Aluminium	Paper
🟨 Net flux	256	31	42	63	61	95	242
🟥 Net flux exc seq	335	31	42	63	70	95	469

The overall impact of diversion to recycling depends strongly on the performance of the landfills from which the waste is diverted. If waste is diverted from a site with a high greenhouse gas flux, then the overall benefits will be proportionately greater. This is illustrated in Figure 26, which shows the effect of diverting waste from sites with limited gas collection.

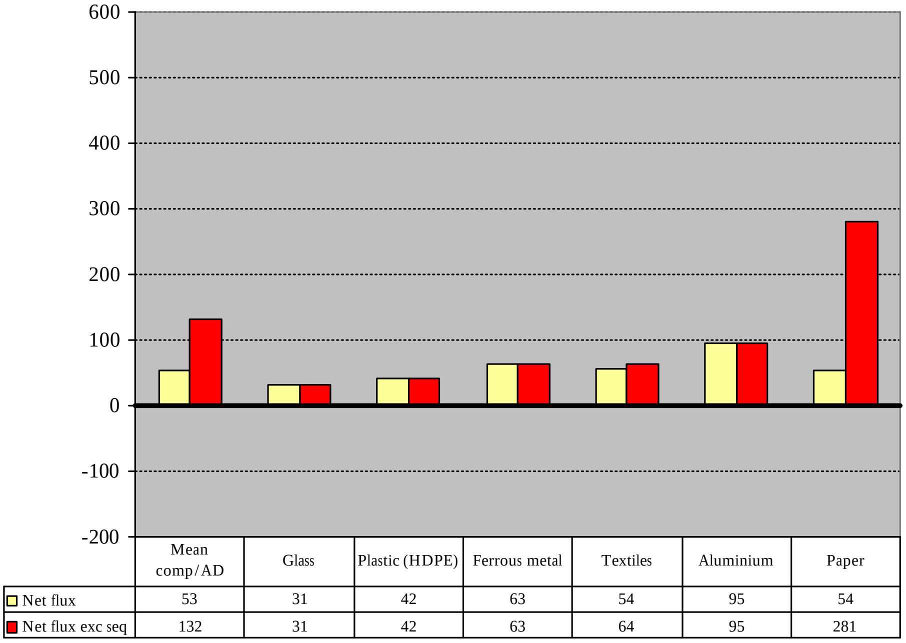
Figure 26: Overall impact of diverting waste components from landfill with limited gas collection to recycling (or composting / AD for putrescible waste), kg CO2 eq/tonne MSW. Positive values indicate a net saving in flux.



	Mean comp/AD	Glass	Plastic (HDPE)	Ferrous metal	Textiles	Aluminium	Paper
🟨 Net flux	317	31	42	63	63	95	298
🟥 Net flux exc seq	395	31	42	63	72	95	525

Where landfill performance is improved through higher standards of gas collection and management, the advantages of recycling in terms of reduced greenhouse gas flux diminish. Figure 27 shows the situation for waste diverted from sites having ‘best practice’ standards of gas management.

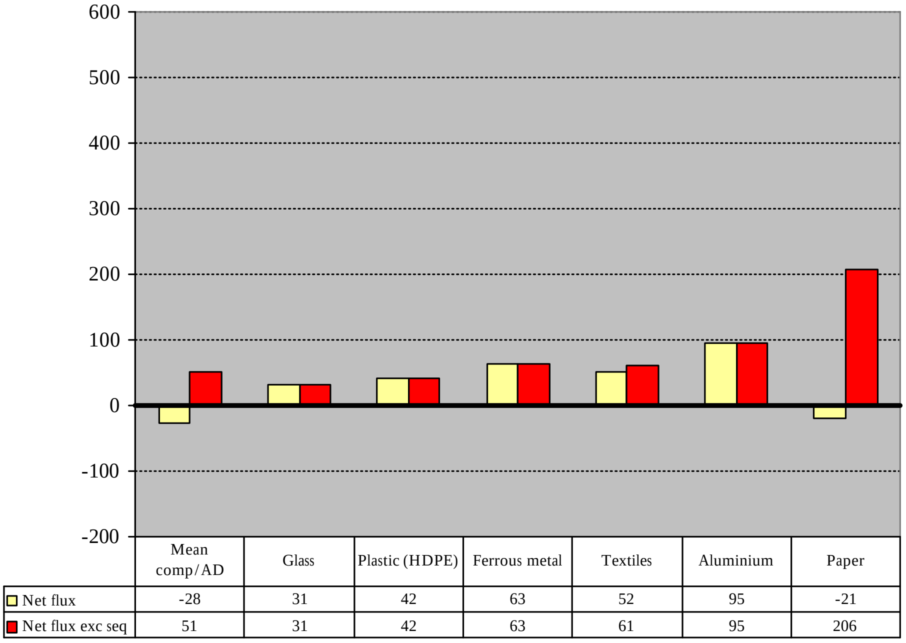
Figure 27: Overall impact of diverting waste components from landfill with best practice gas collection to recycling (or composting / AD for putrescible waste), kg CO2 eq/tonne MSW. Positive values indicate a net saving in flux.



	Mean comp/AD	Glass	Plastic (HDPE)	Ferrous metal	Textiles	Aluminium	Paper
🟨 Net flux	53	31	42	63	54	95	54
🟥 Net flux exc seq	132	31	42	63	64	95	281

Lastly, the impacts of diverting waste from high quality landfills with a restoration layer to further reduce methane emissions is shown in Figure 28. Where the negative flux attributed to carbon sequestration is included, it can be seen that composting or recycling of putrescible or paper results in a small increase (ie a negative saving) in overall greenhouse gas flux, compared with landfill. Where the contribution due to sequestration is excluded, composting and recycling still give positive benefits compared with landfill. In this instance, it appears that although paper recycling still offers the greatest flux saving, a greater benefit is obtained by recycling ferrous metals, aluminium and textiles, than by persuing composting / AD of putrescibles.

Figure 28: Overall impact of diverting waste components from landfill with restoration layers to recycling (or composting / AD for putrescible waste), kg CO2 eq/tonne MSW. Positive values indicate a net saving in flux.



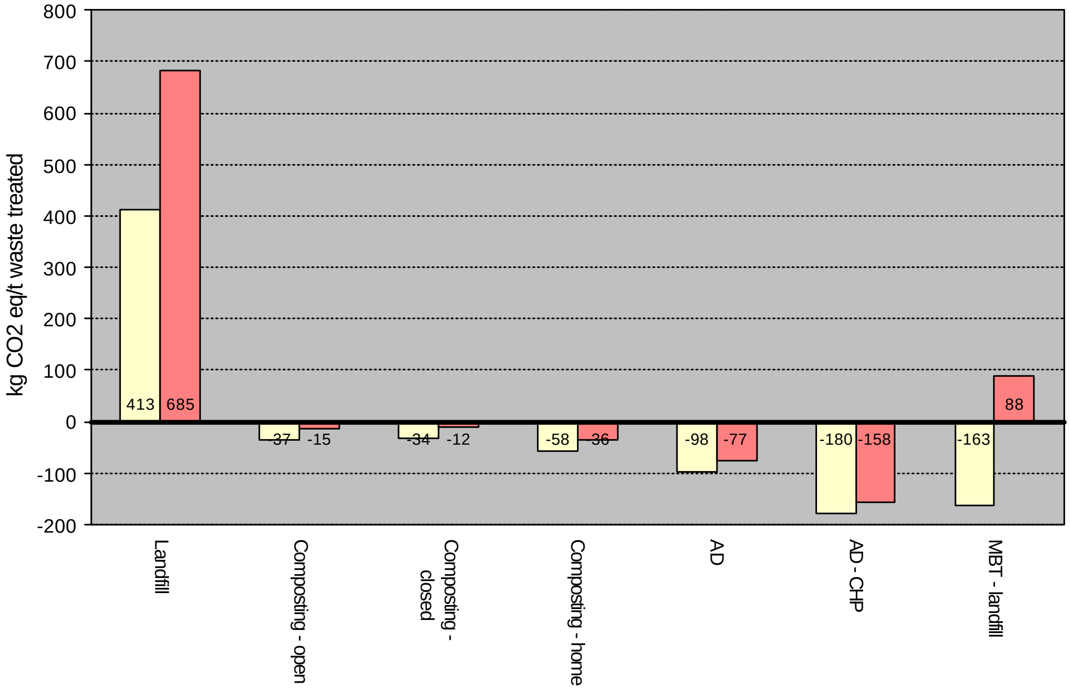
	Mean comp/AD	Glass	Plastic (HDPE)	Ferrous metal	Textiles	Aluminium	Paper
🟨 Net flux	-28	31	42	63	52	95	-21
🟥 Net flux exc seq	51	31	42	63	61	95	206

It is also useful to consider the case where markets for dry recyclable materials are obtained, but opportunities for marketing compost at the price needed to justify separate collection of putrescible wastes have not been found. Contaminated putrescible wastes can then either be landfilled directly or treated through MBT as an alternative to composting / AD. The option of incineration for residual putrescible waste is not considered. Although these wastes are incinerated as part of bulk MSW, higher concentrations are unacceptable for incineration because they reduce the heat value of the waste below the minimum needed for efficient combustion. This comparison is shown in Figure 29. Note that in this case the values are expressed in units per tonne of putrescible waste, rather than MSW. The graph shows the results for EU-average standards of landfilling and EU-2000 average electricity and heat as as the displaced energy sources, and shows the contribution due to carbon sequestration. The average of case 1 and case 2 for MBT has been assumed.

The results show that MBT followed by landfill disposal of the compost performs almost as well in terms of net greenhouse gas flux as AD-CHP of source segregated putrescible waste. MBT performs well because of the credit due to carbon sequestration. On the other hand, the energy benefit of AD with CHP makes this the best option in terms of greenhouse gas flux, ahead of the composting options and AD with energy recovery as electricity only. Composting and AD, as explained above, do not benefit from the large term for carbon sequestration attributed to MBT. However, this evaluation takes no account of non-greenhouse gas advantages of composting over MBT which would have to be taken into account in an overall comparison of the options. If the negative flux due to carbon sequestration is left out of the comparison, the performance of landfill worsens considerably, MBT returns a net *positive* flux and becomes the second worst option, whilst options for source-segregated waste are relatively unaffected.

Figure 29: Net greenhouse gas emissions from options for treating putrescible wastes (base case assumption), kg CO2 eq/tonne putrescible waste.

The left hand bars include the negative flux due to carbon sequestration, the right hand bars exclude this component.



3.2.4 Greenhouse gas fluxes from WEEE management

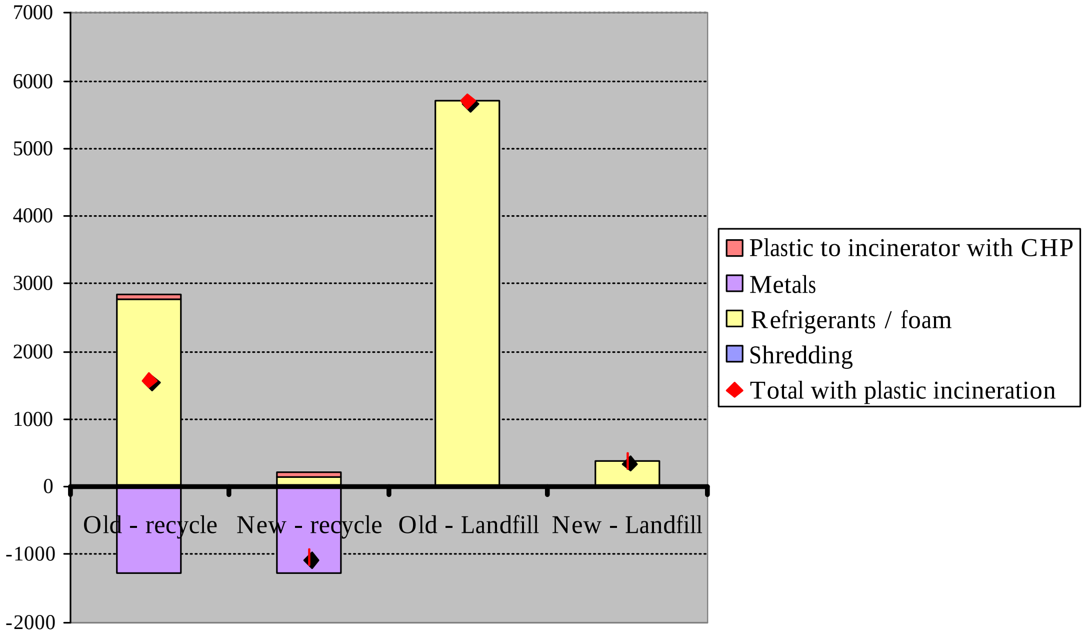
WEEE is estimated to make up about 4% of MSW arisings in Western Europe, but this proportion is increasing [119]. WEEE contains a mixture of ferrous and other metals, plastics, glass etc that reflects the great diversity of products that eventually become WEEE. Further details on this waste stream are given in Appendix 7. Most WEEE is currently landfilled, but forthcoming EU and member state legislation will increase the amount collected for recycling or recovery. Here we compare greenhouse gas emissions from the major elements involved with either recycling or landfilling. Incineration of WEEE is generally not widespread and where it does happen foam and refrigerants are expected to be destroyed by the combustion process.

A major contribution of greenhouse gas emissions from WEEE management comes from refrigerators. Although refrigerators (plus freezers and similar equipment) only make up 9% of

WEEE, the refrigerant itself and the gas used to make the insulating foam have significant global warming impacts. The refrigerant itself is removed for destruction before recycling, but shredding used for metal recycling releases most of the remaining greenhouse gas from the foam into the atmosphere under present practices, as does landfilling. The most potent greenhouse gases found in refrigerators, the CFCs, have been phased out under the Montreal Protocol, since they are also ozone depleting agents. They have been replaced by the less harmful HFCs, which have lower, but still significant, global warming potentials (see Table 4). The analysis presented here considers the greenhouse gas fluxes due to landfilling or recycling of 'old' WEEE, in which the refrigerators contain the now banned CFCs and 'new' WEEE, in which the CFCs are replaced by HFCs. Details of the underpinning assumptions are given in Appendix 7.

The summary results are shown in Figure 30. The greenhouse gas fluxes include those associated with incinerating plastic separated as part of the WEEE recycling process in a mass-burn incinerator with CHP energy recovery. Emissions due to transport and landfill operations are omitted since these are negligible in comparison with the fluxes shown.

Figure 30: Greenhouse gas fluxes associated with WEEE landfilling or recycling (kg CO2 equivalent/tonne). 'Old' refers to WEEE in which the refrigerators contain CFCs, 'New' refers to refrigerators containing HFCs.



The results indicate the major impact of CFCs on greenhouse gas impacts of WEEE management. Landfilling of 'old' WEEE from which CFCs have not been removed makes this materials the most concentrated source of greenhouse gas emissions at almost 5,600 kg CO_2 equivalent / tonne. Even recycling this material creates a net greenhouse gas flux of over 1500 kg CO_2 equivalents/ tonne, due to emissions of CFC from refrigerator foam (the refrigerant is removed for destruction on recycling), unless effective measures are taken to trap emissions. On the other hand, recycling of WEEE containing refrigerators using HFCs ('New WEEE') shows a net benefit in greenhouse gas terms, because the impacts of HFC releases are more than compensated for by the savings due to metal recycling. These benefits are of course absent when new WEEE is landfilled, and releases of HFC from foam and refrigerant contribute to a substantial net greenhouse gas emission of some 390 kg CO_2 equivalents / tonne. Landfilling of WEEE will reduce significantly in the future following the implementation of the WEEE Directive, but agreement is still to be reached over the best way of reducing emissions from refrigerator foam during the crushing and shredding stages of WEEE recycling. Because of the relatively long product lifetime of refrigerators (10-15 years), equipment containing the banned CFCs will be coming through into the waste stream for several years to come.

3.2.5 Costs of waste treatment

The cost of each treatment option is summarised in Table 11. As discussed earlier, these costs are based on gate fees for waste treatment options (or prices paid for recyclable materials at reprocessing plants) plus the incremental costs of source separate collection where appropriate. incremental cost over that of bulk waste

This study has focussed on the assessment of greenhouse gas impacts of waste management options as the primary aim, but indicative data on costs have been collected where possible. A comprehensive assessment of waste management costs throughout the EU would require a detailed study in its own right, although such information would be required for a detailed cost-benefit analysis of the alternative waste management options.

We will consider first the costs of managing bulk MSW, via the treatment options of landfill, incineration and MBT, shown in the top three rows of Table 11. Overall average gate fees (including tax) of 56 Euro/tonne for landfilling and 64 Euro/tonne appear to be comparable, but conceal a very wide range of variation, shown by the ranges in the right hand column. Values towards the lower end of the price range are believed to reflect disposal charges to older or low quality facilities with low environmental protection standards. In the case of landfills, we would expect to see a significant increase in disposal costs as countries bring their landfills into line with the standards required in the landfill directive. In the case of incineration, the disposal fee may be subsidised, through, for example, price support for energy sales. This mechanism is used in the UK, for example, to encourage waste incineration as a ‘renewable’ energy source.

Despite the apparent cost advantage of landfill over incineration, the costs of collecting bulk waste for disposal plays an important role in determining the choice between the two options. Incineration tends to be favoured in large metropolitan areas and cities, where the population density and therefore waste production rates are highest. Incineration is highly capital intensive, and there are significant economies of scale that reduce unit disposal costs at high scales of operation (over about 100 kt/y). In such locations the higher disposal fees for incineration, followed by transportation and disposal of the residues in rural landfills, is often more cost effective than long-haul transport and direct landfilling of raw wastes.

Table 11 Cost of Waste Treatment Options (Euro/t treated)

Treatment	Typical average cost Euro/tonne	Overall range reported Euro/tonne
Bulk MSW		
Landfill	56	11-162
Incineration	64	31-148
MBT	60 - 75	87
Putrescible - food / garden waste		
Separate collection	40 in 2000	0 – 75
	10 in 2020	
Composting		
Open systems	35	16 – 174 (open/closed)
Closed systems	50	0 (home)
Home	0	
AD	65	41-153
Paper		
Separate collection	40 in 2000	
	10 in 2020	
Recycling	-450	
Plastics		
Separate collection	400	250-1242
Recycling	300 (mechanical)	100-500
	150 (feedstock)	50-250
Metals		
Separate collection	40 in 2000	
	10 in 2020	
Recycling		
	-22 (steel)	
	-945 (aluminium)	
Glass		
Separate collection	40 in 2000	
	10 in 2020	
Recycling	-34	-45 – 0
WEEE		
Separate collection	300	200-400
Recycling	50 (large household equipment)	10-80
	250 (refrigerators)	200-300
	450 (equipment with monitors)	100-800
	350 (small household equipment)	200-500

Experience of MBT as a pre-treatment for landfill is almost entirely limited to Germany and Austria at present. The limited disposal fee data collected in this study suggest that the technique has the potential to compete on cost with both landfill and incineration. With limitations due to be implemented on the disposal of untreated biodegradable waste to landfill, MBT appears to offer a solution with little or no increase in cost. Indeed, some further cost reductions could be anticipated through the use of MBT, such as the reduced need for landfill gas and leachate management and better use of landfill void space. Perhaps the greatest barrier at present to the widespread adoption of MBT outside Germany and Austria is lack of familiarity amongst the waste management industry and local authority waste planners.

Composting disposal fees reflect the costs of investment and operation of the facility and income from the sale of the compost so produced. Open systems tend to be cheaper than the more complex closed systems, which may offer better control over process conditions and odours etc. Typical disposal charges of 30 – 50 Euro/tonne for biodegradable wastes processed by composting are less than those for the bulk waste disposal options. In the case of AD, however, costs are higher (~65 Euro/tonne), reflecting the greater complexity of the process. For both composting and AD, we also need to consider the costs of separate collection, which is essential if usable compost is to be produced. In several of the schemes investigated in this study, the costs of separate collection were comparable with the treatment costs, so that overall composting worked out to be more expensive than bulk waste disposal. However, more recent work in which very high participation rates were achieved found that separate collection of biodegradable wastes could be undertaken for little or no increase in cost. This was because the more frequent collection of biodegradable wastes allowed the residue to be collected less often. It is expected, therefore, that with increasing levels of public education and participation, that the incremental costs of separate collection overall will fall substantially in the future. No disposal fees have been shown for home composting, but this does not mean that the option is without costs. Some municipalities may meet either all or part of the costs of providing commercial compost containers to householders who wish to participate, at a cost of about 20-50 Euro each. Alternatively, householders may prefer to construct their own out of recycled timber etc. Home composting costs do not, of course, need to include an element for collection since all the compost is used by the household that produces it.

Recycling costs are highly variable, depending not only on the material in question, but also on market factors such as the price of material made from primary sources, the quality and quantity of material available for recycling and local factors affecting transport costs. Data collected in this study indicates that aluminium recycling in particular generates net income and to a lesser extent, paper recycling also. Recycling of other materials (glass, plastic, ferrous metal, WEEE) from MSW is generally more expensive than the market price paid for the recycled product, and so has to be subsidised

3.3 SCENARIO ANALYSIS

This section presents a scenario analysis of the total greenhouse gas emissions arising for MSW management in the EU in 2020, compared with 2000.

3.3.1 Base data for the year 2000

For these scenarios, and also for the calculations of emissions from a tonne of EU-average waste presented in the previous section, data on the current composition of MSW waste arisings in the EU is taken from the most recent available OECD/EUROSTAT statistics (which are collected via a combined questionnaire) [10]. The shortcomings of this dataset are widely recognised[a], but it is the most complete 'official' set of waste statistics available at the time of writing, and is adequate for present purposes. Greatly improved data should be available shortly as a result of measures to comply with the EC landfill directive, packaging waste directive and the proposed waste statistics directive.

The OECD/EUROSTAT data does not provide a split between food and garden waste, ferrous and non-ferrous metal or textiles and 'other' waste. All of these categories were split based on the ratio between waste components in the UK. We then assume that half of non-ferrous metal waste is aluminium.

The OECD/EUROSTAT data provides recycling statistics only for paper and glass. We have supplemented this data with information from various European trade associations:

- APME for plastics – figures on recycling rates for plastics in MSW in 1998 [11].

- CEPI for paper [12] – figures on overall paper recycling rates in 1998. Around half of paper waste collected for recycling is from MSW. We assume the MSW recycling rate is the same as the overall rate. Note that some of the figures appear unusually large or small due to import or export of paper waste.

- APEAL for steel packaging [13,14]]. We assume that steel packaging comprises 80% of MSW ferrous metal waste (based on a survey of UK dustbin waste [44]) and that the remainder is not currently recycled. In practice some of the packaging waste probably would not be collected for recycling (non-food and beverage cans, e.g. paint cans) but some of the 'other' waste (appliances and household items) probably would be collected. In addition some of the waste not collected separately would be recovered from incinerator ash;

- The European Aluminium Association for aluminium waste [15]. Figures for aluminium can recycling have been used, assuming that 40% of aluminium waste in MSW is cans [16]. This figure comes from the average of three recent surveys of UK dustbin waste. The

---

[a] For example, country data is generally 2-3 years old and so, strictly speaking, represents arisings during the late 1990s, rather than for the year 2000. Returns for some countries are even more out of date (Italy's returns refer to 1985) and for others there are concerns over the reliability of the waste analyses (UK data for putrescible wastes are suspiciously low). There are also other inconsistencies in the data – for example, the total arisings quoted are not always entirely consistent with the sum of the amounts of waste processed by the various treatment options., although such discrepancies are relatively small, especially in comparison with other data quality concerns.

remaining aluminium waste is classed as ‘foil’ (16%) and ‘other’ (42%). ‘Other’ includes discarded appliances and household items (e.g. pans). We assume that foil and ‘other’ waste is not currently recycled. Individual country figures were not used as some of the reported figures appeared anomalously large (e.g. Austria 502%) due to trade flows in aluminium waste. However the figure for the whole of Western Europe was used (41%).

No data was found on textile waste recycling rates. We have assumed a figure of 5% for the whole EU.

OECD/EUROSTAT treatment data does not cover anaerobic digestion or MBT. Also much of the data is out of date (e.g. data for Germany is from 1993). Therefore we have supplemented the data with more recent information where possible.

The assumptions concerning waste arisings, composition and treatment routes for each member state and for the EU as a whole are shown in Table 12. Table 13 on page 56 shows the destinations of various waste components at the EU level.

The scenario analysis is also based on the assumption that sequestered carbon is taken into account in the analysis.

Table 12: Waste arisings, composition and treatment assumptions for EU countries in 2000. Data from OECD 1999 [10] except where otherwise indicated.

		AU	BE	DK	FI	FR	GE	GR	IR	IT	LU	NL	PO	SP	SW	UK	EU
Arisings	kt/y	4110	4852	2951	2100	28800	36976	3900	2032	26605	193	8716	3800	15307	3200	29000	172542
Composition																	
Paper	%	27	16	20	26	25	41	20	33	22	19	27	23	21	44	32	29
Food /garden	%	27	37	47	32	29	23	47	29	43	44	39	35	44	30	21	31
Plastics	%	18	7	5	5	11	3	9	9	7	8	5	12	11.6	7	11	8
Glass	%	8	7	4	6	13	22	5	6	6	7	6	5	7	8	9	11
Metal	%	7	4	2	3	4	8	5	3	3	3	2	3	4	2	8	5
Ferrous	*%*	*5.25*	*3*	*1.5*	*2.25*	*3*	*6*	*3.75*	*2.25*	*2.25*	*2.25*	*1.5*	*2.25*	*3*	*1.5*	*6*	*4*
Non-ferrous	*%*	*1.75*	*1*	*0.5*	*0.75*	*1*	*2*	*1.25*	*0.75*	*0.75*	*0.75*	*0.5*	*0.75*	*1*	*0.5*	*2*	*1*
Textiles / other	%	13	29	24	30	18	3	16	20	19	20	20	23	12.4	9	19	15
Textiles	*%*	*1.37*	*3.05*	*2.53*	*3.16*	*1.89*	*0.32*	*1.68*	*2.11*	*2.00*	*2.11*	*2.11*	*2.42*	*4.80*	*0.95*	*2*	*2*
Other	*%*	*11.63*	*25.95*	*21.47*	*26.84*	*16.11*	*2.68*	*14.32*	*17.89*	*17.00*	*17.89*	*17.89*	*20.58*	*7.60*	*8.05*	*17.00*	*13.18*
Recycling		**AU**	**BE**	**DK**	**FI**	**FR**	**GE**	**GR**	**IR**	**IT**	**LU**	**NL**	**PO**	**SP**	**SW**	**UK**	**EU**
Paper	%	97	18	34	60	43	58	26	13	42	43	52	37	56	82	37	47
Glass	%	88	75	70	62	52	79	26	38	34	52	82	44	37	76	26	52
Ferrous metal	%	60	56	60	12.8	38	64			8	55	62.4	12	18.56	49.6	24	35
Aluminium	%													19	33		16
Plastics	%	8.9	8.6	2.3	0.8	1.5	55	0	1.2	3.7	8.6	3	0.4	1.1	2.5	0.5	7.8
Textiles	%																5
Treatment		**AU**	**BE**	**DK**	**FI**	**FR**	**GE**	**GR**	**IR**	**IT**	**LU**	**NL**	**PO**	**SP**	**SW**	**UK**	**EU**
Landfill	kt/y	887	939	343	1500	9593	18978	3561	1432	24000	94	1768	3610	11758	1200	24000	103663
Incin – no energy recovery	kt/y		207	0	0	2702	0	0	0	0	0	0	0	78	0	700	3687
Incin – with energy recovery	kt/y	431	508	1602	50	7650	6429	0	0	1400	98	2693	0	627	1300	2000	24788
Composting	kt/y	360	428	428	70	1716	2013	3		2501	7	2150	190	2394	100	628	12988
AD	kt/y		67				450					197				2	716
MBT/landfill	kt/y	200					2000										2200
MBT/incineration	kt/y																0

OECD data shows the total amount of waste incinerated together with the proportion going to incinerators either with or without energy recovery. This breakdown is missing in the case of Germany, Italy, Netherlands and Sweden. We have therefore assumed that for these countries all of the incinerated waste goes to plant with energy recovery. UK waste arisings data comes from [45]. See text for details.

Table 13: Waste destinations for EU-15, year 2000.

Treatment options	Units	Whole MSW	Paper	Putres-cibles	Plastics	Glass	Ferrous metal	Non-ferrous	Textiles	Fines	Misc comb	Misc non comb	Fate of MSW
Collected MSW for treatment	%	100	28.9	31.5	8.2	11.2	3.9	1.3	1.9	5.4	6.2	1.6	
Collected MSW for treatment	kt/y	**172,674**	**49,853**	**54,282**	**14,102**	**19,368**	**6,794**	**2,265**	**3,273**	**9,362**	**10,700**	**2,675**	
Recycled	**%**	22	47		8	52	35	8	5				
Recycled	kt/y	37,497	23,514		1,101	10,125	2,407	186	164				22%
Composted	kt/y	12,988	649	12,338									8%
Anaerobic digestion	kt/y	716	72	644									0%
Total removed at source	kt/y	**51,200**	**24,235**	**12,983**	**1,101**	**10,125**	**2,407**	**186**	**164**				**30%**
Remainder	kt/y	**121,473**	**25,617**	**41,299**	**13,001**	**9,243**	**4,387**	**2,079**	**3,109**	**9,362**	**10,700**	**2,675**	
Composition of remainder	%	100%	21%	34%	11%	8%	4%	2%	3%	8%	9%	2%	
Landfill	kt/y	103,663	21,861	35,244	11,095	7,888	3,744	1,774	2,653	7,990	9,131	2,283	60%
Incineration	kt/y	3,687	778	1,253	395	281	133	63	94	284	325	81	2%
Energy recovery	kt/y	24,788	5,228	8,428	2,653	1,886	895	424	634	1,911	2,183	546	14%
MBT/landfill	kt/y	2,200	464	748	235	167	79	38	56	170	194	48	1%
MBT/incineration	kt/y												
Total	kt/y	**185,538**	**52,566**	**58,656**	**15,479**	**20,347**	**7,258**	**2,485**	**3,602**	**10,354**	**11,833**	**2,958**	**108%**

The combined amount of MSW treated by all of the waste management options (shown in the bottom row) is slightly different to the total waste arisings data given in the second from top row because of inconsistencies in the source data from OECD and minor rounding errors. Greenhouse gas fluxes were based on the amount of waste going to each treatment option.

3.3.2 Scenarios for the Year 2020

Four scenarios have been assessed:

1. Landfill Directive Achieved
2. Landfill Directive Delayed
3. Greenhouse Gas Emissions Minimised
4. Biological Treatment Maximised.

The analysis is fairly simple and intended purely to give an illustration of the potential variation in future greenhouse gas emissions depending on scenario. The scenarios involve estimates of the following parameters:

- Changes in waste arisings in the year 2020.
- Changes in fundamental parameters such as incineration efficiencies and landfill gas collection rates in the year 2020.
- Changes in the mix of treatment options employed by each member state.

Changes in waste arisings and basic parameters in the year 2020 are assumed to be broadly the same for each scenario. The scenarios differ from each other mainly in the mix of treatment options used. The only exception to this is that landfill gas control parameters differ for two of the scenarios.

Here we present first the assumptions concerning waste arisings and basic parameters, and secondly the assumptions concerning treatment options for each scenario.

3.3.3 Waste arisings in 2020

Waste arisings in the EU have been growing since records began, with a strong correlation between waste arisings per capita and Gross Domestic Product (GDP) [17]. The EC Waste Management Strategy aims to de-link waste generation from economic activity. However, this aim has not yet been achieved. Between 1990 and 1995 the arisings of waste in the EU grew by 10% while GDP grew by 6.5% [10]. Only the Netherlands and Germany appear to be achieving a reduction in the amount of household waste generated per unit household expenditure [17]. There has also been little progress towards a target to stabilise MSW generation at 300 kg per capita by the year 2000. It seems likely that southern European states will continue to increase waste generation until their levels of economic activity have converged with those of Northern European countries.

Waste statistics even for present day generation of waste are notoriously unreliable [18]. However, there has been one attempt to predict future arisings of household waste based on past trends and predictions of economic growth [19, 20]. The study estimated that household waste arisings would grow by 22% between 1995 and 2010.

For this study we have assumed an annual growth in waste arisings of 2% between 2000 and 2010, followed by a stabilisation of waste arisings between 2010 and 2020. This corresponds with a 22% increase on year 2000 arisings. Although in reality the growth in arisings will differ between countries, with arisings possibly decreasing in some countries, this has not been taken into account in this analysis. The growth in arisings has been apportioned equally between all countries.

We assume no changes in the composition of the waste generated. In reality, certain waste fractions (e.g. plastics) will probably grow relative to other fractions. However this level of analysis is beyond the scope of our study and is unlikely to have a major impact on the results.

3.3.4 Parameter changes in 2020

The following parameters are assumed to have changed in the year 2020:

- Small improvements in incineration efficiencies (see Appendix 3);
- Equal quantities of waste are assumed to go to incinerators with CHP and electricity generation only. All incinerators are assumed to recover energy in some form in 2020, in line with the requirements of the new incineration directive.
- Proportion of paper used in composting and AD schemes (see Appendix 5 and Table 14);
- Landfill gas collection rates – see Table 15;
- MBT composted residue use. All of the compost is assumed to be landfilled, with half of the compost being highly stabilised and half being less well stabilised – greenhouse gas flux will be the mean of Case 1 and Case 2, described in Appendix 4.
- EU-average emissions from electricity generation – assumed to decrease by 10% to reflect an increasing proportion of generation from renewable sources;
- Emissions saved from fertiliser displacement associated with compost use should change in theory (see Appendix 6) – in practice this makes very little difference to the results and so we have omitted to vary this for the scenario analysis.

Table 14 Assumptions concerning the EU average proportion of paper and putrescible waste used in composting and AD schemes in 2000 and 2020

	2000		2020	
	Composting	**AD**	**Composting**	**AD**
Paper	5%	10%	10%	20%
Putrescibles	95%	90%	90%	80%

Table 15: Assumptions concerning landfill gas collection parameters for each scenario

	EU-2000	**Landfill Directive Achieved**	**Landfill Directive Delayed**	**Minimised GHG**	**Maximise biological treatment**
% of landfill sites controlled	68%	95%	80%	95%	95%
gas collection efficiency	54%	60%	54%	70%	60%
% of landfill gas used for energy	60%	60%	60%	60%	60%

3.3.5 Waste treatment assumptions

3.3.5.1 Scenario 1: Landfill Directive Achieved

The landfill directive states that all member states should reduce the amount of biodegradable waste landfilled to 35% of 1995 levels by the year 2016 (or 2020 for states which landfilled more than 80% of their waste in 1995). Biodegradable waste is defined as ‘any waste that is capable of undergoing aerobic or anaerobic decomposition, such as food and garden waste, and paper and paperboard’. In theory this could include a portion of the ‘other’ category and even the biodegradable portion of textile waste. In practice the Directive seems to be interpreted as applying only to food and garden waste and paper. For example, a report by the European Topic Centre for Waste [17] estimates the amount of biodegradable waste landfilled in the EU in 1995 as 55 million tonnes, based on paper and food/garden waste only. We therefore only take paper and food/garden waste into consideration. It is also not yet clear whether composted MBT residue will count as biodegradable waste, or if some allowance will be made for the reduction in degradable carbon content for this waste fraction. Here we assume that MBT is an acceptable alternative to landfilling.

In addition the directive states that all landfill sites should have ‘appropriate’ gas control measures in place, and should use the gas collected for energy generation ‘where possible’, by 2009. However there is no definition of what degree of gas collection is ‘appropriate’ or where energy recovery is ‘possible’. We have assumed that 95% of waste will be disposed of to sites with gas collection in place (making some allowance for sites serving islands or isolated rural communities which can be exempt from the requirements). We assume that the average collection efficiency will increase from our estimated 54% in 2000 to 60% by 2020. The proportion of recovered gas used for energy generation is assumed to be unchanged at 60%, as this is governed largely by the gas generation profile (i.e. some gas is inevitably not utilised when concentrations are too low for energy recovery, before and after the peak gas generation phase).

Several Member States have policies regarding landfilling of waste which have more stringent requirements than those in the Landfill Directive. For example:

- ***Austria:*** no carbon – containing waste after 2004
- ***Belgium:*** only pre treated waste after July 2000 (Flanders only)
- ***Denmark***: only pre treated waste after 1997
- ***Finland***: only pre treated waste after 2005
- ***France:*** only ‘final’ waste after 2002
- ***Germany:*** only pre-treated ‘final’ waste from 1999; no carbon containing wastes from 2005
- ***Italy***: inert and residual wastes only from 2002
- ***Netherlands:*** no combustible wastes to be landfilled after 2001
- ***Sweden:*** no organic material after 2005

We do not take account of these in this scenario, but do so in Scenario 3 (minimise greenhouse gas emissions).

We assume diversion of biodegradable waste to a mix of biological treatment options (i.e. composting, anaerobic digestion, MBT) and incineration. The mix of options varies between countries. Most countries have yet to develop firm plans on the strategy which will be adopted to meet the diversion rates required by the landfill directive. Where information is available on future strategies, then these were used to help define the mix of options, recognising that plans may be subject to change and revision as they develop. Much of the information used was taken from a recent report for the Resource Recovery Forum [21]. For most countries, composting and AD are assumed to grow in line with current trends, and the remainder of the diversion will be to either incineration with energy recovery or MBT.

- In Austria, incineration is a major means of meeting the landfill directive. Maximum use will also be made of MBT. The composting rate is high (14% of MSW) and composting/recycling is growing at a faster rate than MSW arisings.

- Germany is assumed to favour MBT with landfill of the reject fraction. Arisings are actually decreasing and composting/recycling rates are increasing.

- Denmark aims to increase incineration to 50-60% and currently has the highest incineration rate in the EU. The composting rate is low (2%) and composting/recycling is growing only in line with arisings.

- France is assumed to favour incineration. Composting (currently 6%) is growing slower than arisings.

- The Netherlands is assumed to favour incineration although composting will also grow. Separate collection of organic waste is widespread.

- Italy, Spain, the UK, Greece, Portugal and Ireland are highly dependent on landfill. In Spain, arisings are growing faster than recycling and composting. Diversion will come from a mix of composting and incineration.

- In Sweden composting/recycling is growing faster than arisings.

Based on these indications, we define this scenario according to the following criteria:

Recycling is assumed to continue roughly in line with present trends, i.e. modest increases in recycling rates are expected. The rates assumed are shown in Table 16. The figures have been derived as follows.

- **Paper.** CEPI have committed themselves to increasing the EU average recycling rate to 56% by 2005 (CEPI declaration on paper recovery [22]). We take this figure as being a conservative estimate of the baseline rate for 2020. This overall rate is achieved as follows. Countries with low current rates of paper recycling (Belgium, Greece, Ireland) will increase their rates to 30%. Countries with moderate rates (Denmark, France, Italy, Luxembourg, the Netherlands, Portugal, Spain and the UK) will increase their rates to 56% (the CEPI target for 2005). Countries with rates above 56% (Austria, Finland, Germany, Sweden) will maintain their rates at current levels.

- **Glass.** We assume an increase from the current rate of 52% to 60% in 2020. This overall rate is achieved as follows. Countries with low current rates of glass recycling (Spain, Greece, Ireland) will increase their rates to 40%. Italy increases its rate to 45% and the UK to 50%. Countries with moderate rates (France, Luxembourg, Portugal) will increase their rates to 60% (the assumed average). Countries with rates above 60% (Austria, Belgium, Denmark, Finland, Germany, the Netherlands, Sweden) will maintain their rates at current levels.

- **Steel.** We assume that 60% of the recyclable portion of ferrous metal waste (assumed to be 80%, see previous section) will be recovered by 2020. This gives an EU average figure of 48%, compared to 35% in 2000. This overall rate is achieved as follows. Countries with current rates of metal recycling below 48% will increase their rates to 40%. Countries with rates above 48% will maintain their rates at current levels.

- **Aluminium.** We assume that 50% of the recyclable portion of aluminium waste (assumed to be 40%, see previous section) will be recovered by 2020. This gives an EU average figure of 20%, compared to 16% in 2000. We do not differentiate between countries as this would make very little difference to the overall EU emissions.

- **Plastics.** APME estimate that the recycling rate could increase to 11% by 2006 [23], although this rate is for all plastics including industrial sectors where separation and recovery may be easier than for MSW. Another study estimates a ‘business as usual’ rate for mechanical recycling of PVC of 8-9% in 2020, compared to 3% today [26]. The packaging waste directive mandates a recycling rate of 15% for plastics packaging waste (equivalent to around 10% for all plastics) by 2001, but this is unlikely to be achieved across the EU. We assume a modest increase from the present rate of 8% to 10% in 2020.

- **Textiles.** In the absence of any data, we assume a modest increase from our estimate of 5% today to 10% by 2020.

Table 16 Assumptions concerning recycling rates (%) for different scenarios

	EU-2000	Landfill Directive Achieved	Landfill Directive Delayed	Minimised GHG	Maximise biological treatment
Paper	47	56	56	60	56
Glass	52	60	60	75	60
Ferrous metal	35	48	48	72	48
Aluminium	16	20	20	23	20
Plastics	7.8	10	10	20	10
Textiles	5	10	10	25	10

- **Composting and AD.** In countries where composting and /or AD are already established (Austria, Belgium, Denmark, Germany, Italy, the Netherlands, Spain), we assume that 50% of putrescible waste will be collected separately (60% for the Netherlands). 80% of this waste will be diverted to composting and 20% to AD. To this putrescible waste is added paper, which forms 10% of the composting input and 20% of the AD input.

- **Residual waste.** Austria and Germany will maintain their existing levels of incineration and use MBT to pre-treat the remaining waste. Austria will incinerate the MBT reject fraction whereas Germany will landfill it. The other countries will use incineration to make up the rest of the diversion from landfill. All incineration will have energy recovery and 50% is assumed to be with CHP.

The assumptions concerning waste arisings and treatment routes for each member state and for the EU as a whole under scenario 1 are shown in Table 17. Table 18 (on page 63) shows the estimated destination of the various waste components at the EU level.

Table 17: Waste treatment assumptions for Landfill Directive Achieved scenario (scenario 1)

		AU	BE	DK	FI	FR	GE	GR	IR	IT	LU	NL	PO	SP	SW	UK	EU
Arisings	kt/y	5010	5915	3597	2560	35107	45074	4754	2477	32431	235	10625	4632	18659	3901	35351	210328
Recycling																	
Paper	%	90	30	56	60	56	58	30	30	56	56	56	56	56	82	56	56
Glass	%	88	75	70	62	60	79	40	40	45	60	82	60	40	76	50	60
Ferrous metal	%	60	56	60	40	40	64	40	40	40	55	62.4	40	40	49.6	40	48
Aluminium	%	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20
Plastics	%	8.9	8.6	2.3	0.8	1.5	34	0	1.2	3.7	8.6	3	0.4	1.1	2.5	0.5	9
Textiles	%	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10	10
Separate collection																	
Putrescible waste arisings	kt/y	1353	2188	1691	819	10181	10367	2234	718	13945	104	4144	1621	8210	1170	7424	66170
Fraction collected separately	%	50%	50%	50%	25%	25%	50%	25%	25%	50%	25%	60%	25%	50%	25%	25%	41%
Amount collected separately	kt/y	676	1094	845	205	2545	5183	559	180	6973	26	2486	405	4105	293	1856	27431
Treatment																	
Recycled	kt/y	1925	908	668	558	8615	21641	547	363	6297	44	2616	853	3584	1728	9258	59605
Landfill	kt/y	340	385	145	565	3560	7000	1330	550	10220	40	580	1375	4625	440	9350	40505
Incineration	kt/y																0
Energy recovery	kt/y	435	3396	1838	1207	20081	7000	2251	1363	8105	122	4645	1950	5852	1405	14664	74315
Composting	kt/y	595	963	744	180	2240	4561	492	158	6136	23	2188	357	3612	257	1633	24139
Anaerobic digestion	kt/y	162	263	203	49	611	1244	134	43	1673	6	597	97	985	70	445	6583
MBT/landfill	kt/y						3627										3627
MBT/incineration	kt/y	1553															1553
Total	kt/y	5010	5915	3597	2560	35107	45074	4754	2477	32431	235	10625	4632	18659	3901	35351	210328
Biodegradable waste landfilled	kt/y	103	159	71	261	1661	4706	806	309	5003	22	279	656	2237	269	3977	20519
Target for 2020	kt/y	112	165	71	259	1662	4707	807	310	4941	20	298	659	2239	273	3981	**20505**

Table 18: Waste destinations for EU-15, 2020, Landfill Directive Achieved (Scenario 1).

Treatment options	Units	Whole MSW	Paper	Putres-cibles	Plastics	Glass	Ferrous metal	Non-ferrous	Textiles	Fines	Misc comb	Misc non comb	Fate of MSW
Collected MSW for treatment	%	100	28.9	31.5	8.2	11.2	3.9	1.3	1.9	5.4	6.2	1.6	
Collected MSW for treatment	kt/y	210,488	60,770	66,170	17,191	23,610	8,282	2,761	3,990	11,413	13,043	3,261	
Recycled	%	26	56		10	60	48	10	10				
Recycled	kt/y	54,566	34,031		1,719	14,166	3,975	276	399				26%
Composted	kt/y	24,139	1,207	22,932									11%
Anaerobic digestion	kt/y	6,583	658	5,925									3%
Total removed at source	kt/y	85,289	35,896	28,858	1,719	14,166	3,975	276	399				41%
Remainder	kt/y	125,199	24,873	37,312	15,472	9,444	4,306	2,484	3,591	11,413	13,043	3,261	
Composition of remainder	%	100%	20%	30%	12%	8%	3%	2%	3%	9%	10%	3%	
Landfill	kt/y	40,505	8,047	12,071	5,005	3,055	1,393	804	1,162	3,692	4,220	1,055	19%
Incineration	kt/y												
Energy recovery	kt/y	74,315	14,764	22,147	9,184	5,606	2,556	1,475	2,131	6,774	7,742	1,936	35%
MBT/landfill	kt/y	3,627	721	1,081	448	274	125	72	104	331	378	94	2%
MBT/incineration	kt/y	1,553	309	463	192	117	53	31	45	142	162	40	1%
Total	kt/y	205,289	59,428	64,157	16,356	23,101	8,049	2,627	3,796	10,797	12,340	3,085	98%

The combined amount of MSW treated by all of the waste management options (shown in the bottom row) is slightly different to the total waste arisings data given in the second from top row because of inconsistencies in the source data from OECD and minor rounding errors. Greenhouse gas fluxes were based on the amount of waste going to each treatment option. Aluminium is estimated to make up half of the non-ferrous metal component. Assuming that negligible recycling of the remaining non-ferrous metals occurs, the recycling rate for aluminium only is therefore double the rate shown for the combined non-ferrous fraction.

3.3.5.2 Scenario 2: Landfill Directive Delayed

This scenario examines the impact on emissions if the requirements of the Landfill Directive were not fully complied with by the deadlines stipulated in the Directive. We assume that the target to achieve a reduction in biodegradable waste landfilled to 50% of 1995 levels, due to be achieved in 2009 (2013 for countries heavily reliant on landfill), is not actually achieved until 2020. The mix of treatment methods is determined using the same approach as described above, except that only 40% of putrescible waste is assumed to be separately collected for composting and AD in countries with an established composting base, and 15% in other countries. The Netherlands maintains its current rate of 60% and France and Portugal maintain their rates of 20%. Recycling rates are similar to scenario 1.

We also assume that only 80% of landfill sites implement gas control, and that gas collection efficiency remains at our average estimated level for the year 2000 (54%).

The assumptions concerning waste arisings and treatment routes for each member state and for the EU as a whole are shown in Table 19. Table 18 (on page 63) shows the predicted waste destinations for the various waste components under this scenario.

Table 19: Waste treatment assumptions for Landfill Directive Delayed scenario (scenario 2)

		AU	BE	DK	FI	FR	GE	GR	IR	IT	LU	NL	PO	SP	SW	UK	EU
Arisings	kt/y	5010	5915	3597	2560	35107	45074	4754	2477	32431	235	10625	4632	18659	3901	35351	210328
Separate collection																	
Putrescible waste arisings	kt/y	1353	2188	1691	819	10181	10367	2234	718	13945	104	4144	1621	8210	1170	7424	66170
Fraction collected separately	%	40%	40%	40%	15%	20%	40%	15%	15%	40%	15%	60%	20%	40%	15%	15%	33%
Amount collected separately	kt/y	541	875	676	123	2036	4147	335	108	5578	16	2486	324	3284	176	1114	21819
Treatment																	
Recycled	kt/y	1906	878	645	547	8546	21500	517	353	6107	42	2616	842	3473	1712	9157	58842
Landfill	kt/y	430	500	200	760	4950	9730	1830	760	13550	50	900	1920	6100	620	12790	55090
Incineration	kt/y																0
Energy recovery	kt/y	435	3556	1995	1115	19330	7000	2032	1243	6527	125	4325	1507	5408	1372	12157	68127
Composting	kt/y	476	770	595	108	1792	3649	295	95	4909	14	2188	285	2890	154	980	19200
Anaerobic digestion	kt/y	130	210	162	29	489	995	80	26	1339	4	597	78	788	42	267	5236
MBT/landfill	kt/y						2199										2199
MBT/incineration	kt/y	1633															1633
Total	kt/y	5010	5915	3597	2560	35107	45074	4754	2477	32431	235	10625	4632	18659	3901	35351	210328
Biodegradable waste landfilled	kt/y	147	224	106	371	2371	6731	1154	441	7159	29	432	943	3192	394	5684	29379
Target for 2020	kt/y	159	236	102	371	2374	6724	1152	443	7059	29	426	942	3198	390	5688	**29293**

Table 20: Waste destinations for EU-15, 2020, Landfill Directive Delayed (Scenario 2).

Treatment options	Units	Whole MSW	Paper	Putres-cibles	Plastics	Glass	Ferrous metal	Non-ferrous	Textiles	Fines	Misc comb	Misc non comb	Fate of MSW
Collected MSW for treatment	%	100	28.9	31.5	8.2	11.2	3.9	1.3	1.9	5.4	6.2	1.6	
Collected MSW for treatment	kt/y	210,488	60,770	66,170	17,191	23,610	8,282	2,761	3,990	11,413	13,043	3,261	
Recycled	%	26	56		10	60	48	10	10				
Recycled	kt/y	54,566	34,031		1,719	14,166	3,975	276	399				26%
Composted	kt/y	19,200	960	18,240									9%
Anaerobic digestion	kt/y	5,236	524	4,713									2%
Total removed at source	kt/y	79,003	35,515	22,953	1,719	14,166	3,975	276	399				38%
Remainder	kt/y	131,485	25,255	43,216	15,472	9,444	4,306	2,484	3,591	11,413	13,043	3,261	
Composition of remainder	%	100%	19%	33%	12%	7%	3%	2%	3%	9%	10%	2%	
Landfill	kt/y	55,090	10,581	18,107	6,482	3,957	1,804	1,041	1,504	4,782	5,465	1,366	26%
Incineration	kt/y												
Energy recovery	kt/y	68,127	13,086	22,392	8,016	4,893	2,231	1,287	1,860	5,913	6,758	1,690	32%
MBT/landfill	kt/y	2,199	422	723	259	158	72	42	60	191	218	55	1%
MBT/incineration	kt/y	1,633	314	537	192	117	53	31	45	142	162	40	1%
Total	kt/y	206,052	59,604	64,175	16,477	23,174	8,083	2,646	3,824	10,886	12,441	3,110	98%

The combined amount of MSW treated by all of the waste management options (shown in the bottom row) is slightly different to the total waste arisings data given in the second from top row because of inconsistencies in the source data from OECD and minor rounding errors. Greenhouse gas fluxes were based on the amount of waste going to each treatment option. Aluminium is estimated to make up half of the non-ferrous metal component. Assuming that negligible recycling of the remaining non-ferrous metals occurs, the recycling rate for aluminium only is therefore double the rate shown for the combined non-ferrous fraction.

3.3.5.3 Scenario 3: Minimise Greenhouse Gas Emissions

In this scenario we illustrate a theoretical case in which greenhouse gas emissions are minimised. For most inert components, this involves maximising recycling levels, with residual putrescible wastes going to MBT. Our assumptions for each waste fraction are given below.

- **Putrescible (food and garden) waste**. After maximising recycling of the remaining wastes mentioned below, the residual waste is enriched in putrescible wastes, for which the management option giving rise to the lowest greenhouse gas flux is MBT, with landfill of rejects. Incineration is unsuitable for residual wastes enriched in putrescible materials if the gross calorific value of the residue is much below about 7 GJ/tonne.

- **Paper.** We assume that a substantial proportion of paper will continue to be recycled in line with waste management policies. CEPI estimate that some 18.5% of waste paper cannot be recycled, either because it is unrecoverable (e.g. tissue paper etc.) (9.3%), because it is contaminated (e.g. pizza boxes etc.) (2.5%) or because it is out of the waste cycle (e.g. library books etc.) (6.6%). This implies that some 80% could be recycled in future years. However, there will be additional constraints such as the percentage of virgin fibre needed to maintain an adequate fibre length, and the participation rate in collection schemes. We assume that some 60% will be recycled in 2020.

- **Glass.** Recycling is the best option. We assume that 75% of waste glass could be recycled (in Sweden, rates of 84% are being achieved for glass packaging).

- **Ferrous metal.** Recycling is the best option. Ferrous metal is relatively easy to recover, both from source-separate collection and from incinerator ash, due to the possibility of magnetic separation. We assume that 80% of ferrous metal waste could be recovered by 2020.

- **Aluminium.** Recycling is the best option. We assume that 90% of aluminium cans (assumed to be 40% of aluminium waste, see previous section) could be recovered by 2020. A rate of 84% recovery of cans is already being achieved in Sweden. This gives an overall rate of 36% for aluminium waste, compared to 16% in 2000.

- **Plastics.** Recycling is the best option. APME estimate that the overall recycling rate for all sectors could increase to 11% by 2006 [23]. In Sweden, rates of 34% are already being achieved for plastic packaging [24] and in Germany rates of over 50% are being achieved [25]. Packaging accounts for almost 60% of plastics waste [23] so a rate of 50% for packaging equates to an overall rate of 30%. Another report estimates that mechanical recycling of PVC waste could increase from less than 3% today to 18% by 2020 [26]. We assume that the present EU average rate of 8% could be increased to 20% by 2020, although greater increases might be theoretically possible.

- **Textiles.** Recycling is the best option. We have no data, and have assumed a current recycling rate of 5%. Textiles should be relatively easy to collect separately as they are seldom contaminated. Despite the current depressed state of the textile recycling industry (which faces competition from overseas imports of clothing and textiles) we therefore assume that a rate of 25% could be achieved by 2020.

- **‘Other’ waste.** For ‘other’ waste the best option is incineration with CHP, and the second-best is MBT with landfill. However, ‘other’ waste would be contained within the residual fraction for which MBT with landfill is in fact the best option (see above).

The fate of wastes under the Minimise Greenhouse Emissions scenario is modelled at the whole EU level, so we do not show data for individual countries. The estimated waste flows (in ktonnes / year in 2020) are shown in Table 21 on page 69.

3.3.5.4 Scenario 4: Maximise Biological Treatment

Recognising that in reality a number of environmental, economic and social concerns will drive the development of waste management (rather than just the climate change issue) we have also developed a scenario where the use of biological treatment methods is maximised.

We have assumed that 75% of putrescible waste will be collected separately. Two-thirds of this is assumed to be diverted to composting schemes, and one-third to anaerobic digestion schemes with CHP. Recycling rates for paper and inert materials are assumed to be the same as for scenario 1.The remainder of the waste will be treated by MBT with landfill. Clearly it is unlikely that these treatment rates will be implemented across the EU by 2020, and unlikely that all existing incineration capacity would be removed, but this is purely an illustrative scenario.

This scenario is modelled at the whole EU level, so we do not show data for individual countries. The treatment routes are shown in Table 22 on page 70.

Table 21: Waste destinations for EU-15, 2020, Minimise Greenhouse Gas Emissions (Scenario 3).

Treatment options	Units	Whole MSW	Paper	Putres-cibles	Plastics	Glass	Ferrous metal	Non-ferrous	Textiles	Fines	Misc comb	Misc non comb	Fate of MSW
Collected MSW for treatment	%	100	28.9	31.5	8.2	11.2	3.9	1.3	1.9	5.4	6.2	1.6	
Collected MSW for treatment	kt/y	**210,488**	**60,770**	**66,170**	**17,191**	**23,610**	**8,282**	**2,761**	**3,990**	**11,413**	**13,043**	**3,261**	
Recycled	**%**	31	60		20	75	80	18	25				
Recycled	kt/y	65,727	36,462		3,438	17,707	6,625	497	997				31%
Composted	kt/y												
Anaerobic digestion	kt/y												
Total removed at source	kt/y	**65,727**	**36,462**		**3,438**	**17,707**	**6,625**	**497**	**997**				**31%**
Remainder	kt/y	**144,761**	**24,308**	**66,170**	**13,753**	**5,902**	**1,656**	**2,264**	**2,992**	**11,413**	**13,043**	**3,261**	
Composition of remainder	%	100%	17%	46%	10%	4%	1%	2%	2%	8%	9%	2%	
Landfill	kt/y												
Incineration	kt/y												
Energy recovery	kt/y												
MBT/landfill	kt/y	144,332	24,236	65,973	13,712	5,885	1,651	2,257	2,983	11,379	13,004	3,251	69%
MBT/incineration	kt/y												
Total	kt/y	**210,059**	**60,698**	**65,973**	**17,150**	**23,592**	**8,277**	**2,754**	**3,981**	**11,379**	**13,004**	**3,251**	**100%**

The combined amount of MSW treated by all of the waste management options (shown in the bottom row) is slightly different to the total waste arisings data given in the second from top row because of inconsistencies in the source data from OECD and minor rounding errors. Greenhouse gas fluxes were based on the amount of waste going to each treatment option. Aluminium is estimated to make up half of the non-ferrous metal component. Assuming that negligible recycling of the remaining non-ferrous metals occurs, the recycling rate for aluminium only is therefore double the rate shown for the combined non-ferrous fraction.

Table 22: Waste destinations for EU-15, 2020, Maximise Biological Treatment (Scenario 4).

Treatment options	Units	Whole MSW	Paper	Putres-cibles	Plastics	Glass	Ferrous metal	Non-ferrous	Textiles	Fines	Misc comb	Misc non comb	Fate of MSW
Collected MSW for treatment	%	100	28.9	31.5	8.2	11.2	3.9	1.3	1.9	5.4	6.2	1.6	
Collected MSW for treatment	kt/y	210,488	60,770	66,170	17,191	23,610	8,282	2,761	3,990	11,413	13,043	3,261	
Recycled	%	26	56		10	60	48	10	10				
Recycled	kt/y	54,566	34,031		1,719	14,166	3,975	276	399				26%
Composted	kt/y	36,411	1,821	34,591									17%
Anaerobic digestion	kt/y	19,831	1,983	17,848									9%
Total removed at source	kt/y	110,809	37,835	52,439	1,719	14,166	3,975	276	399				53%
Remainder	kt/y	99,680	22,935	13,731	15,472	9,444	4,306	2,484	3,591	11,413	13,043	3,261	
Composition of remainder	%	100%	23%	14%	16%	9%	4%	2%	4%	11%	13%	3%	
Landfill	kt/y												
Incineration	kt/y												
Energy recovery	kt/y												
MBT/landfill	kt/y	91,733	21,107	12,636	14,238	8,691	3,963	2,286	3,304	10,503	12,003	3,001	44%
MBT/incineration	kt/y												
Total	kt/y	202,542	58,942	65,075	15,957	22,857	7,938	2,562	3,703	10,503	12,003	3,001	96%

The combined amount of MSW treated by all of the waste management options (shown in the bottom row) is slightly different to the total waste arisings data given in the second from top row because of inconsistencies in the source data from OECD and minor rounding errors. Greenhouse gas fluxes were based on the amount of waste going to each treatment option. Aluminium is estimated to make up half of the non-ferrous metal component. Assuming that negligible recycling of the remaining non-ferrous metals occurs, the recycling rate for aluminium only is therefore double the rate shown for the combined non-ferrous fraction.

3.3.6 Scenario results

The results of the scenario analysis are shown in Figure 31 (emissions per tonne of EU-average MSW) and Figure 32 (total emissions for the whole of the EU). The figures compare the various treatment options for a tonne of EU-average MSW in the year 2000 with the four scenarios for the year 2020.

For a tonne of whole waste, it is clear that landfill is the worst option, followed by incineration without energy recovery. Incineration with energy recovery provides net benefits due to the displacement of fossil fuels in the energy system. With energy recovery as electricity only, the benefits of energy recovery only just cancel out the direct greenhouse gas emissions from waste combustion, but with CHP there is a significant net benefit. The two MBT options show a large net benefit which arises mainly from carbon sequestration. If carbon sequestration had not been taken into account then these options would have small positive greenhouse gas emissions.

The current EU average mix has positive greenhouse gas impacts of about 50 kg CO_2 equivalent per tonne of waste, significantly lower than the 328 kg CO_2 equivalent per tonne of waste of the 100% landfill case. Although some 60% of waste is currently landfilled, the overall emissions are reduced by credits for energy recovery from incineration and materials recovery from recycling.

All four of the illustrative future scenarios show negative greenhouse gas emissions. The savings from maximising recycling make the ‘minimise greenhouse gas emissions’ scenario the most favourable. The option of maximising biological treatment also shows large emission savings, although this is highly dependent on the credit for carbon sequestration with the assumed widespread use of the MBT option. The two scenarios involving meeting or partially meeting the landfill directive show lesser savings. It must be emphasised that all four of these scenarios involve massive increases in the rate of source separation of organic waste for composting and AD schemes and in the capacity for incineration with energy recovery and/or MBT. It would also be wrong to conclude that the net negative emissions imply that waste arisings need not be minimised. There are much wider implications and impacts, such as the need to minimise use of resources, to reduce harmful emissions from incineration and to reduce the impacts of land use and transport associated with waste management.

Figure 31 Scenario analysis: Greenhouse gas emissions per tonne of MSW

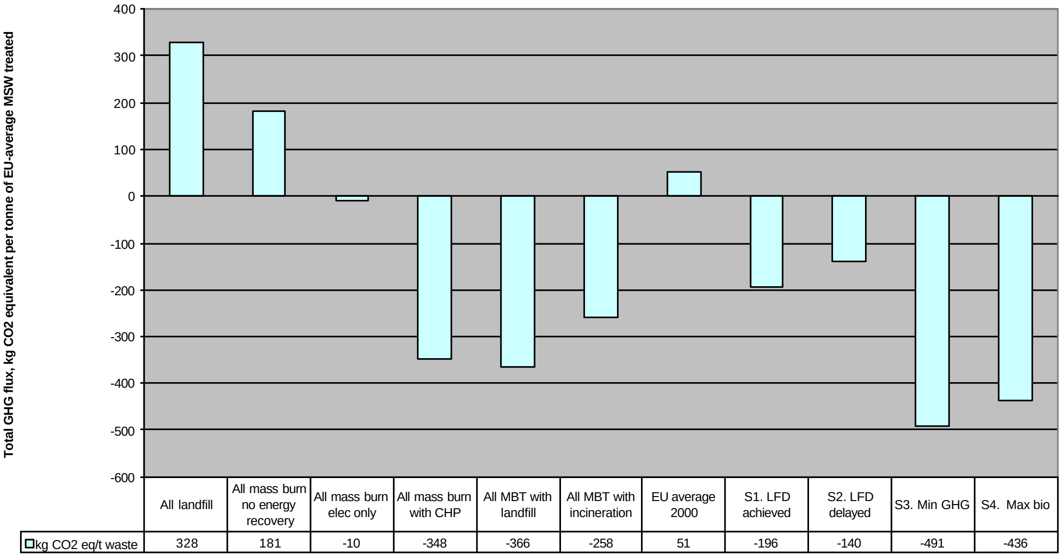
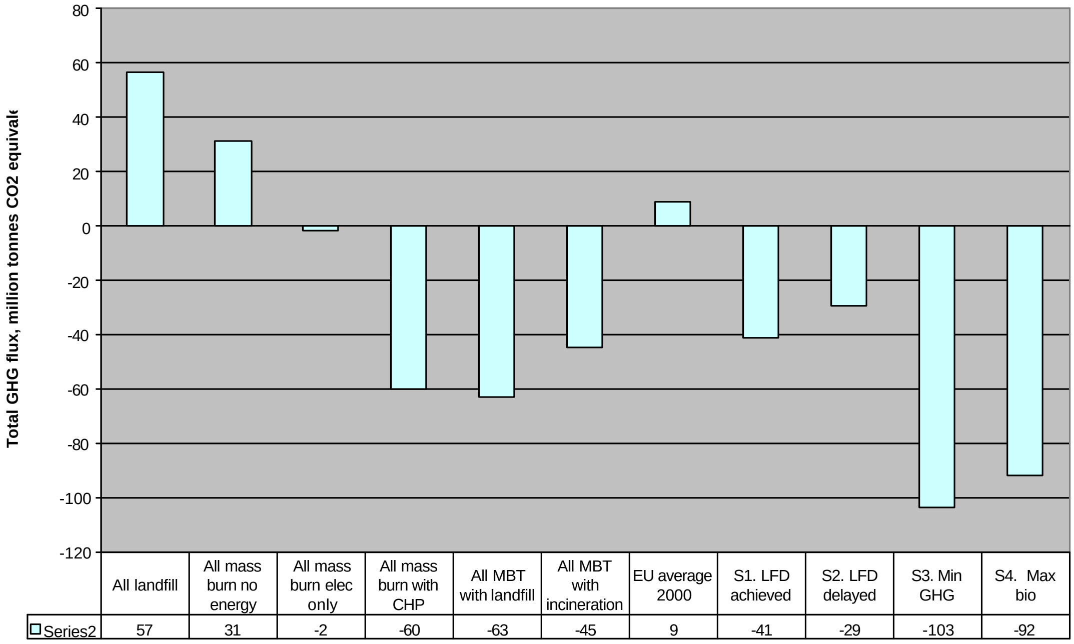


Figure 32 Scenario analysis: Total EU greenhouse gas emissions



4 Conclusions

1. The study has shown that overall, source segregation of MSW followed by recycling (for paper, metals, textiles and plastics) and composting /AD (for putrescible wastes) gives the lowest net flux of greenhouse gases, compared with other options for the treatment of bulk MSW. In comparison with landfilling untreated waste, composting / AD of putrescible wastes and recycling of paper produce the overall greatest reduction in net flux of greenhouse gases. The largest contribution to this effect is the avoidance of emissions from landfills as a result of recycling these materials. Diversion of putrescible wastes or paper to composting or recycling from landfills operated to EU-average gas management standards decreases the net greenhouse gas flux by about 260 to 470 kg CO_2 eq/tonne of MSW, depending on whether or not the negative flux credited to carbon sequestration is included.

2. The issue of carbon sequestration is a particularly important for landfills (and for MBT compost after landfilling), where the anaerobic conditions enhance the storage of carbon. Carbon sequestration plays a relatively small role in the overall greenhouse gas flux attributed to composting, because of the relatively rapid rate of decomposition of the compost after its application to (aerobic) soils.

3. The advantages of paper recycling and composting over landfilling depend on the efficiency with which the landfill is assumed to control landfill gas emissions. For sites with only limited gas collection, the benefits of paper recycling and composting are greater, but less when best practice gas control is implemented. In this case the net greenhouse gas savings from recycling and composting range from about 50 to 280 kg CO_2 eq/tonne MSW. If landfills further reduce methane emissions with a restoration layer to enhance methane oxidation, then recycling and composting incur a small net penalty, increasing net greenhouse gas fluxes to about 20-30 kg CO_2 eq/tonne MSW, if carbon sequestration is taken into account. If sequestration is neglected, then recycling and composting attract a net flux saving of about 50 (putrescibles) to 200 (paper) kg CO_2 eq/tonne MSW.

4. The study has also evaluated the treatment of contaminated putrescible waste using MBT, which may be appropriate if such waste cannot be obtained at high enough quality for composting with the aim of using the compost as a soil conditioner. MBT performed almost as well as AD with CHP in terms of net greenhouse gas flux from putrescible waste, but this advantage was largely determined by the credit for carbon sequestration. If this was not taken into account, then composting or AD of source-segregated wastes remained the best options. Omitting carbon sequestration significantly worsens the greenhouse gas fluxes calculated for landfills and MBT, but has a much smaller effect on composting or AD.

5. It must be emphasised that the apparent advantage of high-quality landfilling over composting and recycling of putrescibles and paper noted above refers only to greenhouse gas fluxes. Issues of resource use efficiency, avoided impacts due to paper making from virgin pulp and improvements in soil stability, fertility and moisture-retaining properties stemming from the use of compost in agriculture must all be considered as part of the assessment of the overall 'best' option. These factors are outside the remit of the present study, but their inclusion would almost certainly point to recycling and composting in preference to any form of landfill disposal for these waste components. Improving landfill

gas management to reduce greenhouse gas emissions is therefore essentially an ‘end of pipe’ solution, which reduces only one of the impacts of landfilling biodegradable waste without tackling the root cause.

6. For other materials (glass, plastics, ferrous metal, textiles and aluminium), recycling offers overall net greenhouse gas flux savings of between about 30 (for glass) and 95 (for aluminium) kg CO_2 eq/tonne MSW, compared with landfilling untreated waste. For these materials, the benefits are essentially independent of landfill standards and carbon sequestration.

7. For mainstream options for dealing with bulk MSW as pre-treatment for landfill, the option producing the lowest greenhouse gas flux (a negative flux of some 340 kg CO_2 eq/tonne MSW) is MBT (including metals recovery for recycling) with landfilling of the rejects and stabilised compost. MBT with incineration of rejects (energy recovered as electricity) gives a smaller net negative flux of about 230 kg CO_2 eq/tonne. Mass-burn incineration where half the plants operate in electricity only and half in CHP mode gives a net negative flux of about 180 kg CO_2 eq/tonne MSW. If all the incineration capacity were assumed to operate in CHP mode, then the net flux from incineration would be almost the same as from MBT with landfill of rejects. On the other hand energy recovery from incineration as electricity only would produce a net flux of only –10 kg CO_2 eq/tonne. These figures are based on EU-average landfill gas control, inclusion of carbon sequestered in MBT compost after landfilling and the replacement of electricity and heat from EU-average plant mix.

8. If the benefits of carbon sequestration are left out of the comparison of options just presented, then the MBT options both produce net positive greenhouse gas fluxes of 23 to 55 kg CO_2 eq/tonne MSW. Incineration is unaffected by assumptions on carbon sequestration.

9. The performance of MBT with landfilling of rejects is further improved as higher standards of landfill gas control are implemented, relative to mass-burn incineration, provided the contribution from carbon sequestration is included. If sequestration is omitted, incineration continues to perform better than MBT.

10. As stated in point 7 above, under the baseline assumptions used in this study, MBT with landfill of rejects gives rise to a lower (net negative) greenhouse gas flux than MBT with incineration of rejects. The main reason for this difference is lies in the source of greenhouse gas emissions in the two options. In MBT with landfill, methane emissions from the landfilled material is the main contributor to the positive flux, whilst for MBT with incineration, methane emissions are much lower but are more than outweighed by fossil carbon dioxide released from incinerating the plastic rejects. The relative performance of the two options depends crucially on the effectiveness of landfill gas control and, in the case of MBT with incineration, the energy source that is displaced by recovering energy from incineration. In the analysis performed here, we have assumed that electricity only is recovered, although in some cases there may be opportunities for recovering heat as well. This would further enhance the performance of MBT with incineration compared with MBT with landfill. It appears therefore that the choice between these options will largely depend on local circumstances, although either will offer a major improvement over current practices of landfilling untreated bulk MSW.

11. The issue of the source of displaced energy is critical to the performance of incineration in terms of net greenhouse gas flux. The base case is predicated on the assumption that energy from waste displaces electricity or heat generated at a CO_2 emission factor representative of average EU power and heat sources. For electricity, there has been an increasing trend to combined cycle gas turbine technology in recent years, but this has not been assessed separately because the emission factor from this technology is very close to average plant mix. Two alternatives to replacement of 'average' electricity are considered. They are (a) replacement of coal-fired power generation, and (b) replacement of electricity generated from renewable sources – in this case wind. The example given in (a) could come about, for example, from the accelerated retirement of an old coal-burning power station due to the commissioning of new incineration capacity, or through the use of RDF as a coal substitute. Example (b) may result from the inclusion of energy from waste (ie incineration) technology within a member state's target for renewable energy – as is the case in the UK. The greater the CO_2 emission factor of the replaced generation source, the greater the emission saved due to its replacement by incineration.

12. Replacement of coal-fired electricity generating plant by mass-burn incineration would result in a net negative greenhouse gas flux of almost 400 kg CO_2 eq/tonne MSW, with equal proportions of power only and CHP incineration capacity. Under these circumstances, mass-burn incineration would give practically the same emission saving as recycling and composting of source segregated materials. With all incinerators in CHP mode, mass-burn incineration would be the best overall option in terms of greenhouse gas flux. Combustion of RDF as a coal substitute in power stations or cement kilns gives rise to a net negative greenhouse gas flux of about half this sum.

13. A different picture emerges for the situation in which the electricity displaced by incineration comes from wind power, as an example of low-emissions renewable energy sources. Here the displaced generation source has almost no greenhouse gas emissions. In this case, mass-burn incineration is virtually neutral in greenhouse gas terms. In comparison, MBT with landfill of rejects produces a net negative flux of almost 340 kg CO_2 eq/tonne MSW, which makes it the best option for non-source segregated wastes. MBT with incineration of rejects gives a net negative flux of about 150 kg CO_2 eq/tonne MSW. These comparisons are on the basis of sequestered carbon being included in the overall flux from the MBT options.

14. If carbon sequestration is omitted, incineration and MBT with landfill of rejects have a similar net greenhouse gas flux in absolute terms (of 8 to 26 kg CO_2 eq/tonne MSW), whilst that for MBT with incineration is much higher, at about 135 kg CO_2 eq/tonne MSW.

15. Alternatives to mass-burn incineration have also been evaluated. From the perspective of greenhouse gas fluxes, emissions from pyrolysis and gasification are assessed as being similar to those of mass-burn incineration. Greenhouse gas fluxes from RDF manufacture and combustion (plus landfill of residues and recycling of recovered metals) depends highly on the fuel which they replace. Combustion as a replacement for average electricity plant mix results in higher greenhouse gas fluxes than for mass-burn incineration, due mostly to methane emissions from the landfilled residue left over from RDF manufacture. Improvements in landfill site gas control therefore improve the performance of this option

relative to mass-burn incineration, although overall this RDF option performs consistently worse in greenhouse gas flux than MBT with incineration of rejects.

16. Recycling of WEEE containing CFC refrigerants and foam agents now banned because of their ozone –depleting properties results in a net increase in greenhouse gas flux due to the escape of some of these agents during recycling operations. This leakage is more than sufficient to compensate for the considerable greenhouse gas benefits of recycling the metals from WEEE. Nevertheless, recycling of WEEE containing these materials is far preferable to landfill, where the greenhouse gas flux would be much higher. The use of less harmful refrigerants and foam agents and the adoption of more efficient collection procedures will largely eliminate the net positive greenhouse gas flux associated with WEEE recycling and result in substantial net greenhouse gas savings, due largely to the avoided emissions attributable to metal recycling. However, a considerable backlog of equipment containing CFCs remains to come through to the waste stream over the next 5-10 years and further efforts to minimise the release of GHG during recycling would be desirable.

17. Overall, emissions of greenhouse gas associated with transportation of waste, residues and recovered materials are small in comparison with the much larger greenhouse gas fluxes in the system, such as those related to avoided energy / materials, landfill gas emissions and carbon sequestration. Variations in emissions due to alternative assumptions about transport routes and modalities will therefore have a negligible impact on the overall greenhouse gas fluxes of the waste management options.

18. The study has evaluated four scenarios alternative scenarios of waste management in the year 2020 and compared the impacts on greenhouse gas fluxes with the year 2000. Achievement of the landfill directive’s target to reduce the landfilling of untreated wastes in 2016 to 35% of 1995 levels is predicted to result in an overall reduction in greenhouse gas flux from a positive flux of 50 kg CO_2 eq/tonne in 2000 to a negative flux of almost 200 kg CO_2 /tonne in 2020. Even if achievement of the directive’s target is delayed until 2020 (rather than 2016), then a negative flux of about 140 kg CO_2 eq/tonne results. Further reductions in greenhouse gas fluxes (to about –490 kg/CO_2 /tonne) could be achieved through investment in recycling, incineration with CHP and MBT. Alternatively, a scenario with no incineration and maximum biological treatment of waste achieves an overall greenhouse gas flux of –440 kg CO_2 eq/tonne.

19. The study has also examined the costs of waste disposal through the various waste management options, as reflected in disposal fees or the prices commanded by recycled materials. Wide difference in disposal costs exist between different member states. Landfill disposal, currently the cheapest option, will inevitably increase in cost with the requirement for higher environmental standards and the consumption of void space as existing sites fill up and close. Little information is available on the costs of MBT, but what there is suggests that this option may become increasingly competitive with landfill and incineration, especially when benefits of increased efficiency of landfill void space use and lower requirements for gas and leachate control are taken into account. Further growth in composting and AD for food and garden wastes will depend to a large extent on continuing success in reducing the costs of separate collection of feedstock and in establishing local markets for the compost product. Recycling remains highly dependent on the market value of the recycled product. With the principal exception of aluminium, the price of materials recovered from MSW does not cover the costs of separating and reprocessing, compared with virgin materials, and

such operations usually require subsidy. This is particularly so of plastic wastes. In this instance the option of co-incineration as a coal-replacement offers comparable greenhouse gas benefits to recycling but at a substantially lower cost.

20. Overall, the study finds that source-segregation of various waste components from MSW, followed by recycling or composting or AD of putrescibles offers the lowest net flux of greenhouse gases under assumed baseline conditions. Improved gas management at landfills can do much to reduce the greenhouse gas flux from the landfilling of bulk MSW, but this option remains essentially an ‘end of pipe’ solution. Incineration with energy recovery (especially as CHP) provides a net saving in greenhouse gas emissions from bulk MSW incineration, but the robustness of this option depends crucially on the energy source replaced. MBT offers significant advantages over landfilling of bulk MSW or contaminated putrescible wastes in terms of net greenhouse gas flux.

21. It must be emphasised that in practice other impacts of waste management options will need to be considered in addition to just greenhouse gas fluxes. These wider considerations will include factors such as resource use efficiency (which will, for example, impinge upon the choice between the *disposal* option of MBT and the *recycling* option of composting or AD) and the impacts of other emissions such as those associated with waste incineration. Furthermore, substantial environmental benefits are associated with the use of compost to improve soil organic matter status and more environmentally-benign methods of cultivation, but only the relatively modest benefits associated specifically with greenhouse gas fluxes have been considered in this study.

Appendices

Appendix 1. Mobilisation

WASTE COLLECTION AND TRANSPORT

Vehicle exhaust emissions of carbon dioxide arise during collection of waste from households and transport onwards to the various treatment plants. Nitrous oxide (N_2O) is also emitted from vehicle engines, but this has a minor impact[a]. The model includes a variety of routes for collection and transport, depending on the waste stream and the treatment option. Collection methods include:

- kerbside collection, either of unsorted waste or source-separated waste
- delivery by the householder to a household waste[b] site
- delivery of recyclables to a 'bring' site such as bottle and can banks at supermarkets.

Collected waste is either taken directly to a treatment or sorting plant or via a refuse transfer station. The collection and transport routes taken for each treatment option are summarised in Figure A1.33. Further details are given for each waste management option in the following sections.

The emissions generated during transport depend on the vehicle type, size of payload and distance travelled. Assumptions regarding vehicle types are given in Table A1.23 and the data used in the model for each transport stage is summarised in Table A1.24 below.

Table A1.23 Vehicle type assumptions

Vehicle type	Payload (t)	Average emissions (kg CO_2/km)
Car (medium sized gasoline)	0.01	0.21 (now) 0.14 (2020)
Small lorry (L1) 3.5-7.5 t rigid	5	0.45
Large lorry (L2) <33t articulated	20	0.84
Refuse collection vehicle (RCV)	6.67	0.71

Source: [27]

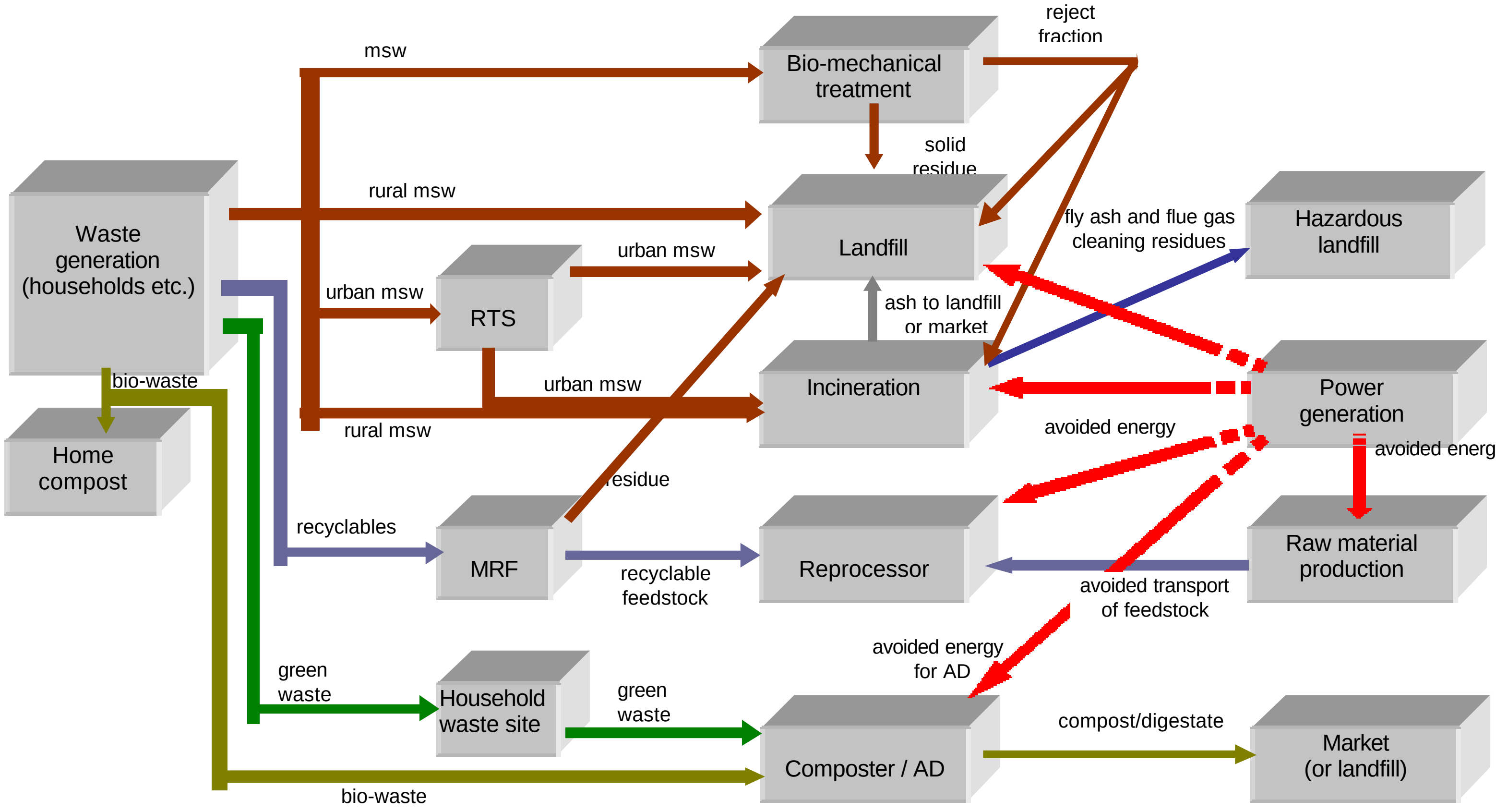
Collection frequency varies across Europe. Waste may be collected daily in Southern European as opposed to weekly or twice weekly in Northern European Countries, as the higher temperatures mean that food waste begins to degrade quickly. However, collection frequency is assumed to have no impact on emissions, as there will be no change in the total weight of waste collected. There would only be an impact if a high collection frequency implied that collection vehicles would not be filled, so that the emissions per tonne of waste collected would increase. It is assumed that local waste collection authorities will optimise their collection systems to avoid the use of half-empty vehicles.

---

[a] N_2O emissions from a diesel heavy goods vehicle are only 1% of CO_2 emissions from the vehicle (typically 9 g CO_2 eq/km of N2O compared to 770 g CO_2/km). (Source: Revised 1996 IPCC Guidelines for national Greenhouse Gas Inventories: Reference Manual).

[b] Household waste sites are facilities for householders to deposit bulky wastes and recyclables that are not collected by the collection service. They were formerly known as civic amenity sites in the UK.

Figure A1.33: Details of treatment routes



Source separate collection

There is an increasing recognition of the benefits of collecting waste in separate fractions, allowing easy diversion of glass, metal and paper for recycling, and biodegradable waste for composting or anaerobic digestion. It is possible to separate mixed waste mechanically or manually at a centralised sorting plant, and this may be successful for recovery of dry recyclables, but it is not possible to obtain uncontaminated biodegradable waste this way. The success of composting and anaerobic digestion plants depends heavily on production of good quality usable compost and this requires source-separated kitchen and garden waste.

There are a variety of methods of source separate collection. The first schemes to be introduced were 'bring' systems which require the householder to deliver materials to a household waste site or to bottle, can and paper banks at a supermarket or other convenient location. However, higher collection rates can be achieved with kerbside collection and this is now increasing. This requires householders to place kitchen and garden waste or dry recyclables in a separate box, bag or bin. Some schemes collect mixed dry recyclables which are then separated either by the collection staff as they load the materials into the collection vehicle, or at a centralised materials reclamation facility (MRF). Other schemes ask the householder to sort the waste into different compartments or containers. Some schemes collect separated waste simultaneously with residual waste, whereas others make separate trips using different vehicles.

In general, separate collection of waste should not give rise to greater emissions from vehicles, if the waste collection system is optimised as discussed above so that all refuse collection vehicles operate at full loads. However, there are some exceptions to this rule. The assumptions concerning additional emissions (if any) arising from source separate collection of waste are discussed in the separate sections on recycling, composting and anaerobic digestion.

Separate collection of waste has an important effect on the awareness of householders of the impacts of the waste that they create. This can lead to reduction of waste at source but this effect has not been taken into account in this analysis.

COLLECTION COSTS

Broadly speaking collection costs are common to all treatment options. Therefore we have not included an estimate of the costs of collecting bulk waste from households. However, as discussed above, the composting, anaerobic digestion and recycling options depend to some extent on separate collection of waste. We have therefore estimated the incremental costs of separate collection schemes compared to bulk waste collection.

In theory there is no reason why source separate collection should be more expensive than bulk collection of unsorted waste, other than the small initial capital cost of providing extra containers and publicity for the scheme. Separate collection of recyclables and biodegradable waste will reduce the quantity of residual waste, allowing savings to be made which should offset any additional collection rounds. However, because most of the experience so far has been with small, pilot schemes some very high costs have been reported. High costs arise mainly if participation rates and collection rates are low, so that the cost of the scheme appears high compared to the volume of collected waste. Because schemes have been operating only for a few years, the costs of the initial capital investment in containers and publicity also appear high compared to the volume of collected waste. Costs are particularly significant for recyclable plastics, because of their very low density and the need for a high degree of sorting.

Nevertheless, some schemes have reported very low or zero extra costs. These tend to be schemes where good publicity has ensured a high participation rate. A recent study in Italy [28] showed that introducing intensive door to door collection of food waste from individual households did not necessarily lead to an increase in collection and transport costs. The high participation and yield rate which the door to door collection encouraged meant that the percentage of food waste in the residual waste was reduced, and residual waste could be collected less frequently, offsetting the costs of collecting the food wastes. Other studies suggest higher costs. For example, a Dutch study suggested an increase from 45 Euro per tonne for normal mixed waste collection to between 50 and 75 Euro per tonne for source separated organics collection, i.e. an increase of 5 to 25 Euro per tonne[29]. Unpublished estimates in the UK suggest that collection costs rise from 27 to 40 Euro per tonne to 33-54 Euro per tonne, an increase of 6 to 14 Euro per tonne. A figure of 11 Euro per tonne was used in a recent UK study[30]. As these are costs per tonne of collected waste, and the biodegradable fraction collected is only about 30% of this, the cost per tonne of collected organic waste is about 3 times these values, i.e. 15 to 75 Euro/t. Estimates of transport and collection costs for compostable waste have also been obtained for Italy (70-150 Euro/t), the Netherlands (55Euro/t) and Spain (30-60Euro/t)[31] but it is not clear whether these are absolute or incremental costs.

Although costs appear high for many schemes at present, we expect these costs to reduce over time as kerbside collection becomes more widespread and participation and collection rates increase. An incremental cost of 40 Euro per tonne for kitchen and garden waste and dry recyclables except plastics appears to be a reasonable average at present, perhaps reducing to 10 Euro per tonne for the 2020 scenarios.

Reported estimates for plastics collection and sorting costs vary widely. The costs depend on the degree of sorting necessary. Less sorting is required for chemical (feedstock) recycling than for mechanical recycling. A recent UK study of PVC recycling [32] reports an estimate of Euro250 per tonne for mixed plastics waste for feedstock recycling and Euro1110 per tonne for mechanical recycling of bottles. This compares well with a 1992 UK report of the cost of collecting plastics for HDPE mechanical recycling [33] which estimates Euro90 for collection, Euro90 for sorting, Euro40 for baling and Euro28 for transport, giving a total of Euro248 per tonne. A report on plastics recycling in Germany [34] reports that costs of Euro1242 per tonne were borne by the German body responsible for collection and sorting of plastics packaging waste in 1996, though these costs were expected to reduce by up to 50% by 2000. Figures from RECOUP[35], the UK organisation specialising in plastic container recycling give the costs of plastic bottle collection in the range Euro280 to 310 per tonne to get plastic bottles to the reprocessor. RECOUP have also estimated that the costs of collecting and delivering plastic bottles to a MRF are in the range Euro110 to 140 per tonne [36].

For this study we have taken an incremental cost of Euro1000 per tonne for mechanical recycling and Euro250 per tonne for feedstock recycling.

The costs of onward transport of collected waste are not include in the model as they are insignificant compared to collection and treatment costs. The only exception to this is that the cost of transporting plastics to the recycling plant is included in the estimate of collection costs due to the low density of plastics.

Table A1.24: Transport data used in the model

Treatment	Stage	Type	From	To	Payload type	Vehicle Type	Average Payload (t)	t payload per t waste	km	kgCO_2/km	kgCO_2/t	% by route	Weighted Emission Factor kgCO_2/t
Landfill	Collection	Direct	Household	Landfill	MSW	RCV	6.67	1.00	40.0	0.7	4.3	19%	0.8
		Via RTS	Household	RTS	MSW	RCV	6.67	1.00	40.0	0.7	4.3	81%	3.5
		Via RTS	RTS	Landfill	MSW	L1	5	1.00	40.0	0.4	3.6	81%	2.9
Total													**7.2**
Incineration	Collection	Kerbside direct	Household	Incin	MSW	RCV	6.67	1.00	40.0	0.7	4.3	19%	0.8
		Via RTS	Household	RTS	MSW	RCV	6.67	1.00	40.0	0.7	4.3	81%	3.5
		Via RTS	RTS	Incin	MSW	L1	5	1.00	40.0	0.4	3.6	81%	2.9
	Fly ash and flue gas cleaning residues to hazardous landfill		Incin	hazardous landfill	Ash and FG residues	L1	5	0.04	100.0	0.4	0.4	100%	0.4
	Ash to market		Incin	market	Ash	L2	20	0.30	13.0	0.8	0.2	100%	0.2
	Unmarketable ash to landfill		Incin	Landfill	Ash	L1	5	0.30	40.0	0.4	1.1	0%	0.0
	Metals to reprocessor		Incin	Metals reprocessor	Metal scrap	L2	20	0.07	100.0	0.8	0.3	100%	0.3
Total													**7.7**
MBT	Collection	Kerbside	Household	MBT plant	MSW	RCV	6.67	1.00	40.0	0.7	4.3	100%	4.3
	Compost to market		MBT plant	Market	Composted residue	L2	20	0.30	10.0	0.8	0.1	0%	0.0
	Unmarketable compost to landfill		MBT plant	Landfill	Composted residue	L1	5	0.30	0.0	0.4	0.0	100%	0.0
	Reject to landfill		MBT plant	Landfill	Reject fraction	L1	5	0.20	0.0	0.4	0.0	91%	0.0
	Reject to incinerator		MBT plant	Incin plant	Reject fraction	L1	5	0.20	40.0	0.4	0.7	9%	0.1
	Metals to reprocessor		MBT plant	Metals reprocessor	Scrap metal	L2	20	0.03	100.0	0.8	0.1	100%	0.1
Total													**4.5**
Composting/ AD	Collection	Kerbside	Household	Composter / AD plant	Putrescibles	L1	5	1.00	40.0	0.4	3.6	67%	2.4
		HHW site	Household	HHW site	Putrescibles	CAR	0.08	1.00	5.0	0.2	12.8	33%	4.2
		HHW site	HHW site	Composter / AD plant	Putrescibles	L2	20	1.00	40.0	0.8	1.7	33%	0.6
		Home compost	Household	Compost heap	Putrescibles			1.00	0.0	0.0	0.0	0%	0.0
	Residue to landfill		Composter / AD plant	Landfill	Residue	L1	5	0.10	40.0	0.4	0.4	100%	0.4
	Compost to market		Composter / AD plant	Market	Compost	L2	20	0.40	10.0	0.8	0.2	100%	0.2
	Unmarketable compost to landfill		Composter / AD plant	Landfill	Compost	L1	5	0.40	40.0	0.4	1.4	0%	0.0
Total													**7.7**

RTS=refuse transfer station

Table A1.24: Transport data used in the model - continued

Treatment	Stage	Type	From	To	Payload type	Vehicle Type	Average Payload (t)	t payload per t waste	km	kgCO2/km	kgCO2/t	% by route	Weighted Emission Factor kgCO2/t
Recycling	Collection	Bring	Collection point	MRF	Mixed recyclables	L1	5	1.00	40.0	0.4	3.6	40%	1.4
		HHW site	Household	HHW site	Mixed recyclables	car	0.08	1.00	5.0	0.2	12.8	20%	2.6
		HHW site	HHW site	MRF	Mixed recyclables	L1	5	1.00	40.0	0.4	3.6	20%	0.7
		Kerbside	Household	MRF	Mixed recyclables	L1	5	1.00	40.0	0.4	3.6	40%	1.4
		Kerbside	Household	MRF	Plastics	L1	1	1.00	40.0	0.4	17.8	40%	7.1
		Take back	Household	retailer	WEEE	car	0.08	1.00	0.0	0.2	0.0	40%	0.0
		HHW site	Household	HHW site	WEEE	car	0.08	1.00	5.0	0.2	12.8	60%	7.7
	Residue to landfill		MRF	Landfill	Residue	L1	5	0.05	40.0	0.4	0.2	100%	0.2
	MRF to reprocessor	Paper	MRF	Paper reprocessor	Paper	L2	20	0.95	100.0	0.8	4.0	100%	4.0
		Glass	MRF	Glass reprocessor	Glass cullet	L2	20	0.95	100.0	0.8	4.0	100%	4.0
		Ferrous metal	MRF	Ferrous metal reprocessor	Ferrous scrap	L2	20	0.95	100.0	0.8	4.0	100%	4.0
		Aluminium	MRF	Aluminium reprocessor	Aluminium scrap	L2	20	0.95	100.0	0.8	4.0	100%	4.0
		Plastic	MRF	Plastic reprocessor	Plastic feedstock	L2	17	0.95	100.0	0.8	4.7	100%	4.7
	Collection point to reprocessor	Textiles	Collection point	Textile reprocessor	Textiles	L2	20	0.95	100.0	0.8	4.0	100%	4.0
	HHW site etc to reprocessor	WEEE	HHW site etc	WEEE reprocessor	WEEE	L2	20	1.00	100.0	0.8	4.2	100%	4.2
Total		Paper											**10.31**
		Glass											**10.31**
		Ferrous metal											**10.31**
		Aluminium											**10.31**
		Plastic											**15.29**
		Textiles											**10.31**
		WEEE											**12.08**

Appendix 2. Landfill

LANDFILL PROCESS DESCRIPTION

Landfill can be defined as the *managed* disposal of waste on land. As such, it is distinguished from *dumping*, which is characterised by the absence of control of the disposal operations and lack of management of the dump site. Waste dumping still occurs in some less-developed parts of the EU, particularly in some southern states, but is being phased out.

There is a huge variation in the characteristics of managed landfills across the EU, and also within member states. At one end of the scale are small, shallow sites with minimal control on the type or quantity of waste entering and no gas collection or leachate management. At the other are large, deep sites with multiple liners where the waste is monitored, compacted and covered, gas is collected for flaring or energy use and leachate is collected and treated to prevent groundwater pollution. Consequently there is wide variation in greenhouse gas fluxes from landfills of different qualities and also a wide variation in disposal fees, associated with the costs of environmental protection measures. Disposal costs are due to rise further, with the implementation of the landfill directive. This will, amongst other things, require most landfills accepting biodegradable wastes to implement gas collection by 2009 (or 2013 for states currently highly dependent on landfill).

Landfill Greenhouse Gas Emissions

In a modern landfill site, decaying wastes use up the oxygen entrained within the waste mass, creating anaerobic conditions. The depths of wastes typically employed means that oxygen is used up faster than it can diffuse in from the air. Under anaerobic conditions, the waste continues to degrade to produce landfill gas, which contains roughly 50% methane and 50% carbon dioxide (see Box 1). The carbon dioxide is assumed to be all short-cycle as only biogenic materials will degrade[a]. In sites with no gas control, the gas migrates to the surface of the landfill site and is released. In sites with gas control, a low permeability cover prevents gas release and a system of wells and pumps is used to extract the gas. The collected gas is either flared or combusted for energy recovery – in either case it is converted to short-term carbon dioxide. Uncollected gas migrates through the cover, or restoration layer if present, where some of it is oxidised to carbon dioxide. Some gas may also escape through cracks or imperfections in the surface layers and around the edge of the landfill. The pathways involved are shown in Figure A2.35.

---

[a] Fossil-derived carbon in waste is assumed to be entirely in non biodegradable polymers in plastic and textiles and other minor waste components. Some fossil-derived organic substances in MSW are in reality biodegradable, such as certain solvents and other household chemicals, but as these make up such a small proportion of MSW they have been omitted from the analysis. Biodegradable plastics, which are beginning to gain applications as waste sacks, are made from short-cycle carbon sources. They have not been considered in the analysis.

Box 1: Evolution of Landfill Gas

The composition of landfill gas changes as the waste decomposes. Eight distinct phases have been recognised [37]. These are as follows:

- **Phase 1.** An **aerobic phase** following waste deposition in which the residual oxygen is used up. This phase typically lasts for a few days to months, depending on local factors such as temperature and moisture availability.
- **Phase II.** **Acid phase**. Populations of facultative and fermentative anaerobic bacteria develop, producing volatile fatty (aliphatic) acids, CO_2 and H_2, displacing the remaining N_2 entrained with the waste. Phase II may last for weeks to years, depending on conditions.
- **Phase III.** **Initial methanogenic phase**. Microbial respiration reduces oxygen concentrations to extremely low values, allowing populations of methanogenic bacteria to develop, producing CH_4. Concentrations of H_2 and CO_2 start to fall.
- **Phase IV.** The **stable methanogenic phase**. Here the remaining H_2 is used up in the reduction of CO_2 to CH_4 and H_2O. Phase V may begin within months to years after waste deposition and last for decades. Landfill sites which collect gas for energy recovery are often designed around an assumed life time of 10 – 15 years of phase IV. Typical landfill gas collected in this phase consists of 40 – 65 % by volume of CH_4, with most of the balance made up by CO_2 . A vast range of trace components are also present (such as volatile fatty acids, reduced sulphur compounds etc) plus water vapour at saturation point. These substances usually make up only 1 or 2 % of the landfill gas, but account for its characteristic sweetish smell.
- **Phase V.** **Air intrusion**. The rate of methanogenic activity begins to fall as substrate is used up, resulting in air beginning to enter the waste. Lower rates of gas formation lead to relatively faster washout of CO_2, so that its concentration falls relative to that of CH_4.
- **Phase VI**. **Methane oxidation**. Rates of methanogenisis have now fallen to low levels, allowing the rate of air ingress to increase, so that the surface layers of the waste and the capping material now become aerobic. Methane migrating through these layers is increasingly oxidised to CO_2 by methanotrophic bacteria. Methane concentration in the gas decreases, whilst that of CO_2 steadily increases.
- **Phase VII**. **CO_2 phase**. Return of aerobic conditions. By now the rate of landfill gas formation has virtually ceased because of substrate limitation and anaerobic decomposition becomes inhibited by the ingress of O_2 in the air. This allow the aerobic decomposition of solid organic matter resistant to anaerobic decomposition.
- **Phase VIII**. **Soil air phase**. The final phase occurs when degradable organic matter has been oxidised and the landfill gas resembles that of typical soil air.

The duration of each of these phases is highly variable. Apart from the initial aerobic decomposition, which may be complete in days to months, the remaining phases have durations measured in years, decades, or even centuries for the final phases. The changes in landfill gas composition over the eight phases described above is illustrated in Figure A2.34

Figure A2.34: Changes in landfill gas composition.

Re-drawn from Christiansen & Kjeldsen [37]. The durations of the eight phases are not drawn to scale.

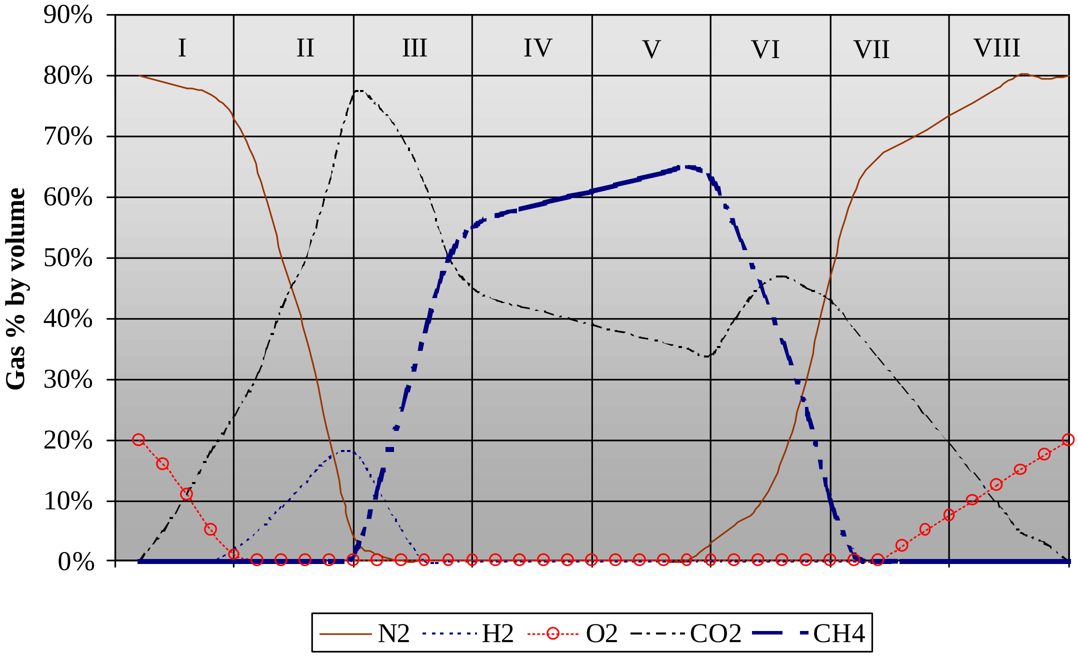
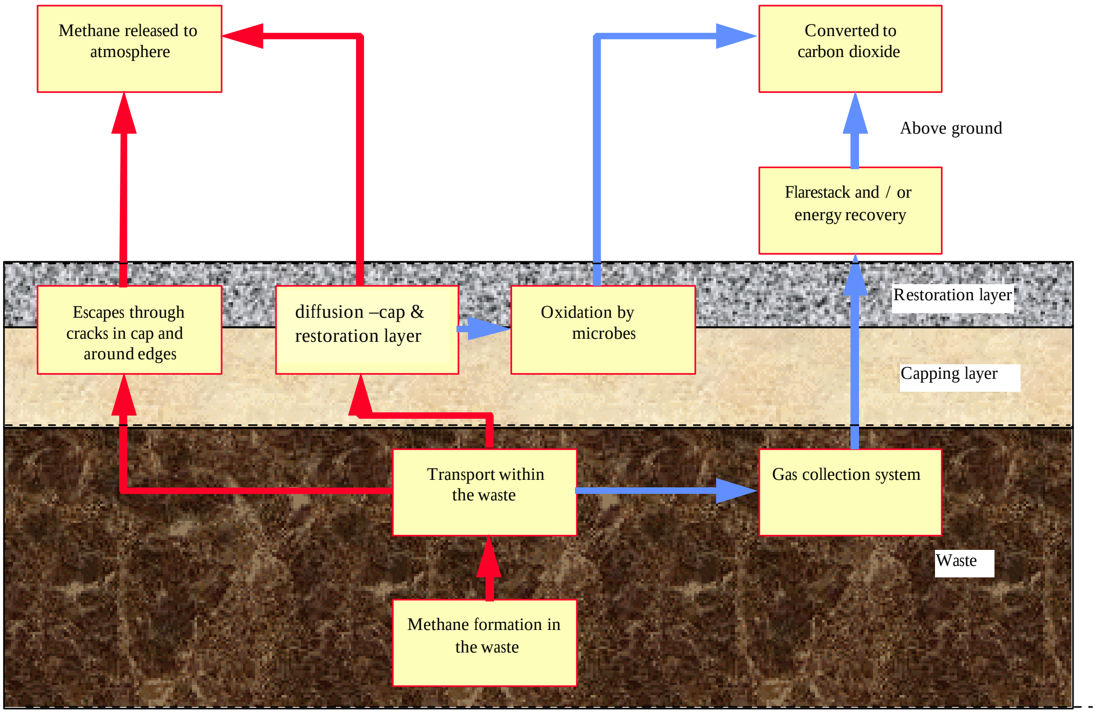
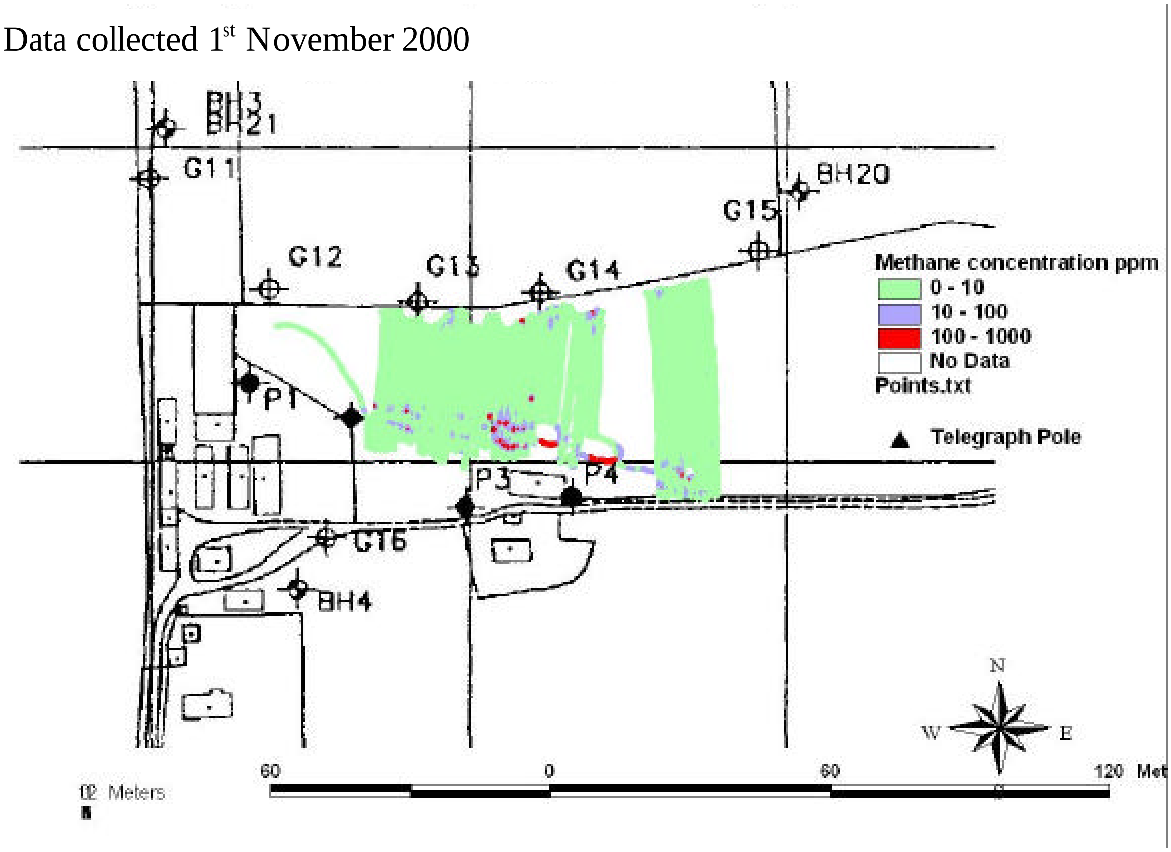


Figure A2.35: Pathways of methane in landfills [38]



Estimates of methane fluxes from landfills are subject to a great deal of uncertainty. One of the major difficulties comes from the highly heterogeneous pattern of emissions across the landfill surface. Small cracks in the capping layer or areas where the cap has not been properly applied can represent major leakage routes, even though such areas make up a very small proportion of the total area of the site. Because of the high rates of gas flow through these cracks, there is very little opportunity for soil microbes to oxidise the methane in these high-flow pathways. On the other hand, methane-oxidising bacteria are abundant in landfill cover material and are usually thought to be capable of consuming nearly all of the methane that diffuses through the bulk of the cover material. The problem comes when the methane short-circuits the microbes living in pores in the cover material and instead flows through cracks and fissures. Average rates of methane oxidation are therefore very hard to predict and subject to a great deal of site to site variation. The problem of ‘hot spots’ of methane emissions is clearly illustrated in Figure A2.36, where the red spots show the location of high methane concentrations, which follow a crack along the southern edge of a landfill surface.

Figure A2.36: Map of methane emission hotspots on a landfill shown using GIS software. Data were collected using AEA Technology's Groundhog™ system.


Data collected 1st November 2000

In addition to CH_4, small amounts of N_2O may also be released from landfills and landfill gas combustion. These emissions are considered to be too small to make a significant contribution to greenhouse gas fluxes from landfills and have therefore been omitted from further analysis. A brief summary of the rationale behind this decision is given in Box 2.

Box 2: N_2O emissions from landfills

N_2O is a product of bacterial metabolism and combustion processes, and so it is necessary to ask if waste decomposing in landfills or emissions from landfill gas flares and engines need to be taken into account in the analysis.

Estimated N_2O emissions for the EU-15 have been reported at 1050 kt in 1994, of which agriculture was the major source at 484 kt. Some 13 kt was attributed to waste, primarily through emissions from waste incineration.

Nitrous oxide emissions from landfills have rarely been measured. Bognor et al[39] reports maximum emissions of 9×10^{-3} g m^{-2} d^{-1} for landfills in the USA compared with CH_4 emissions up to five orders of magnitude higher. A Swedish study is also cited that reported fluxes of 0.8 g m^{-2} d^{-1} from a landfill with a mineral cover soil, but 2.7 g m^{-2} d^{-1} for a cover soil containing sewage sludge with a high N content. These values fall within the wide range of values reported for fertilised and unfertilised natural soils. Bognor concludes that conventional landfills are unlikely to be major contributors to atmospheric nitrous oxide. Consequently direct emissions of N_2O from landfills have been excluded from the present study.

Stationary combustion sources account for just over 12% of EU-15 N_2O emissions in 1994, of which waste combustion is a significant source [**3**]. Further discussion of this source of N_2O is provided in the section dealing with waste combustion. The issue here is to quantify likely emissions from landfill gas combustion, in flares and engines. IPCC [40] quotes an emission factor for gas-fired stationary combustion source of 0.1 g N_2O / GJ of fuel. One GJ of CH_4 emitted to the atmosphere from a landfill would have a global warming impact equivalent to 394 kg of CO_2 . If the same quantity were burnt in an engine or flarestack, the N_2O emitted would have a global warming impact equivalent to 0.031 kg of CO_2[a]. Even allowing for the uncertainty in the emission factor, it seems unlikely that N_2O from landfill gas combustion will make a significant contribution to greenhouse gas impacts of landfill. We have therefore omitted further consideration of this impact route.

The following steps are considered in assessing the greenhouse gas emissions of landfilling of untreated MSW:

- **Mobilisation** of the waste – collection from households and household waste sites and delivery to the landfill;
- **Treatment** – emissions associated with processing the waste at the landfill;
- **Disposal** – emissions arising from waste disposed of in landfills and effects of carbon sequestration;
- **Displaced emissions** from energy recovery from landfill gas

Mobilisation emissions

Waste is collected direct from households and taken either straight to the landfill site (in rural areas) or via a refuse transfer station (RTS) in urban areas. The proportion of the population living in urban areas (81% in the EU [41]) determines the fraction of waste using each route.

Transport routes:

Rural: Household → landfill
Urban: Household → Refuse transfer station (RTS) →landfill

---

[a] Assuming a CV for methane of 38 MJ/Nm^3, 1 GJ would be equivalent to 26.3 Nm^3, or 1.17 kmoles. Methane has a molecular mass of 16, so the mass of methane would be 18.8 kg. This is equivalent to 394 kg CO_2 , assuming a GWP for methane of 21. If 1 GJ of methane were combusted, this would produce 0.1 g of N_2O. Taking a GWP for N_2O of 310 gives a global warming impact equivalent to 0.031 kg of CO_2 .

Vehicle emission characteristics and assumed journey distances are given in Appendix 1. We assume here that 89% of the untreated MSW is collected from households (HH) and taken waste transfer stations (RTS) using a refuse collection vehicle (RCV). After compaction at the RTS, the waste is taken by large lorry (L2) to the landfill. The remaining 11% of MSW is taken directly from the household to the landfill by the RCV. Estimated transport emissions from these routes are shown in Table A2.25. Total emissions are estimated at 7.2 kg CO_2 /tonne of MSW. Handling of waste at the WTS will contribute further CO_2 emissions due to the use of diesel and electricity for handling the waste, but this is considered to be less than 1 kg CO_2 /tonne, and so has been left out of the calculations.

Table A2.25: Mobilisation emissions associated with landfilling untreated MSW.

From	To	Load	Vehicle	Per cent of waste via route	CO_2 emissions, kg/tonne of waste
HH	WTS	MSW	RCV	89%	0.8
RTS	Landfill	MSW	L2	89%	3.5
HH	Landfill	MSW	RCV	11%	2.9
Total					**7.2**

Treatment emissions at the landfill

Operations at the landfill site involve the following steps:

- Weighing the waste on the delivery vehicle as it enters the site;
- Waste is taken to the working area and tipped out;
- Waste is then spread and compacted using a bulldozer or landfill compactor;
- Daily cover of soil or clay is moved to the working area at the end of each day;
- Daily cover is spread and compacted;
- Final cover material is delivered, spread and compacted after the working area reached the desired waste depth.
- Electricity is used for operating leachate and gas collection pumps but this is often generated on site by diesel generators.

The fuel used in each of these steps results in CO_2 emission. The most energy intensive step is the spreading and compaction of each load of waste. The amount of fuel used depends very much on local site practices. For this analysis we have used figures from a large landfill site in the UK which uses 975,000 litres of diesel fuel per year and handles 2.2 million tonnes of waste [42]. This equates to 1.2 kg CO_2 per tonne of waste. Energy usage for waste handling at the RTS is much less intensive, so it is justified to exclude that element from mobilisation emissions.

Disposal emissions

In landfills, waste degrades and gas is released slowly over time. Over a period of 100 years, which is the time scale typically used for greenhouse gas studies, most of the landfill gas will have been released from landfilled waste. Therefore it is valid to estimate emissions per tonne of waste landfilled using the default IPCC methodology[43], which treats methane emissions as though they take place instantaneously after the waste is landfilled. This is a reasonable approximation in cases where the quantities and composition of the waste are reasonably constant over time.

The key parameters are:

- ***Degradable organic carbon content (DOC)*:** the fraction of the waste made up of biodegradable carbon;
- ***Dissimilable DOC (DDOC)*:** the fraction of the DOC that dissimilates (ie mineralises) to CO_2 and/or CH_4. The remainder is assumed not to degrade to gaseous products under landfill conditions within the 100-year horizon.
- ***Methane content:*** the average CH_4 content by volume in the biogas. The remainder is assumed to be entirely CO_2. We assumed that 50% of the carbon was released as methane and 50% as carbon dioxide. This is in line with the IPCC default values.

About 10 – 20% of carbon in waste going to landfills is incorporated into dissolved organic matter in the landfill liquor (leachate). This carbon subsequently mineralises to carbon dioxide and / or methane, depending on the conditions. The IPCC approach does not explicitly distinguish between methane emitted directly from waste or from the subsequent breakdown of leachate carbon, and both sources are included within the overall sum.

DOC

Estimates of DOC have been derived from estimates of the total carbon content of the waste, together with estimates of the proportion of this total carbon which is biogenic and therefore degradable.

Estimates for DOC are shown in Table A2.26. The first column shows total carbon contents, based on a detailed chemical analysis of household waste [44]. The second column shows the proportion of this carbon assumed to be degradable. For paper, food and garden waste, the whole carbon content was assumed to be degradable. For textiles, we assumed that half the waste was of biogenic origin (e.g. cotton, wool) and the rest synthetic and therefore not biodegradable. The third column shows the DOC, i.e. the product of the first two columns.

We have compared these figures to estimates of the DOC of a tonne of whole mixed waste, using an average EU composition for MSW based on OECD figures [10]. Putrescible waste was assumed to be 55% food waste and 45% garden waste, based on UK data[a]. The split of the OECD category ‘textiles and other’ into the sub-categories of textiles, fines and miscellaneous combustible waste was also based on UK data [45]. Column 4 of the table shows the proportion of each component in a tonne of EU-average MSW, and column 5 shows the proportion of DOC contributed by each component to the whole MSW. The sum of these values (0.18) is consistent with the range of estimates of total DOC in MSW from developed countries of North America, Oceania and Western Europe and Scandinavia, quoted by IPCC as 0.08 to 0.19. The breadth of this range is an indication of the uncertainties experienced in quantifying emissions from the waste sector.

DDOC

---

[a] Food waste is all collected from the kerbside whereas some garden waste is delivered to household waste sites. The UK Digest of Environmental Statistics shows approximately 4mt waste collected from HHW sites. Brown et al (99) implies that about half of this is putrescible waste, i.e. garden waste, making 2mt. Kerbside collected waste in the UK is about 20mt of which 20% is putrescible i.e. 4 mt. 83% of this is food waste and 17% garden waste according to a UK dustbin analysis (NHWAP) i.e. 3.3mt food 0.7 mt garden. So the total is 2.7 mt garden waste to 3.3 mt food waste which is a split of 45% to 55%.

Unfortunately, the proportion of DOC which is dissimilable under landfill conditions is very poorly known. DDOC represents two fractions of carbon:

1. organic carbon which is resistant to degradation in landfill conditions, made up mostly of lignin which does not degrade under anaerobic conditions.
2. organic carbon which is theoretically degradable but which does not degrade over the 100 year time scale, perhaps due to factors such as limited access to air, water or nutrients.

The problem is that the degradation rates of individual waste components in a real landfill site are very poorly characterised. There have been a number of laboratory studies of degrading waste, which report a very wide range of estimates and measurements for gas evolution of both whole MSW and its constituents. Stegmann et al [46] in a recent review of methods for predicting landfill gas emissions, quote gas production rates in the literature ranging from 50 to over 400 Nm^3 / tonne of MSW. Choosing 'representative' values is therefore problematic. Also, laboratory conditions are not necessarily comparable to conditions in a real landfill. There has been one US study examining waste in-situ in landfills [47]. This reported that around three quarters of food waste and half of garden waste appeared to have degraded over a period of 25 years. Paper appeared to have hardly degraded over this period – indeed, the layers of waste in the landfill site were dated by reading the dates on newspapers in the waste! However, US landfills are generally drier than European landfills and therefore degradation would not necessarily be comparable.

Estimates for DDOC are shown in Table A2.26. Column 6 shows the proportion of DOC assumed to be dissimilable. Based on the observations in US landfills, we assumed that 75% of the degradable carbon would actually be released during landfill of food waste, 50% for garden waste and 35% for paper. Column 7 shows the resultant estimate of DDOC (i.e. DOC multiplied by the dissimilable fraction of DOC). Assuming 50% of the dissimilable carbon was released as methane and 50% as carbon dioxide, columns 8 and 9 show figures for kilograms of carbon dioxide and methane produced per tonne of waste stream treated.

Our estimates of gas production were then compared with experimental data from two recent studies involving lysimeter experiments in which gas emissions from MSW were measured. Stegmann et al [46] quote a mean value of 120 m^3 of biogas per tonne of German MSW on a fresh weight basis, with a range of 100 – 180 m^3 /tonne. Barlaz et al [reported in 67] quote 147 m^3/tonne of US MSW. Our estimates of 149 m^3 /tonne of MSW are consistent with these experimental results.

To explore the sensitivity of the results to the assumptions regarding dissimilable DOC, we have also tested ranges of values for the two most significant waste fractions: paper and putrescible waste. The ranges tested are shown in column 6 of Table A2.26.

Table A2.26: Estimates of short-cycle carbon dioxide and methane generated from landfilled waste

	1	2	3	4	5	6	7	8	9	10
	total carbon content (TC) of waste component	**proportion of TC which is degradable**	**degradable organic carbon (DOC) as % of waste component**	**composition of EU-average MSW**	**contribution of each component to DOC of a tonne of MSW**	**% of DOC which is dissimilable *(sensitivity range)***	**dissimilable organic carbon (DDOC)**	**methane generated kg CH_4/t**	**CO_2 generated kg CO_2/t**	**carbon sequestered kg CO_2/t**
	(a)	(b)	(c=a*b)	(d)	(e=c*d)	(f)	(g=c*f)	(h=g*50%* 16000/12)	(j=g*50%* 44000/12)	k=(c-g) *44000/12
Paper	33%	100%	33%	29%	9.6%	35% *(20%-50%)*	12%	77	212	786
Food	*15%*	*100%*	*15%*	*17%*	*2.5%*	*75%*	*11%*	*73*	*202*	*135*
Garden waste	*24%*	*100%*	*24%*	*14%*	*3.4%*	*50%*	*12%*	*80*	*220*	*440*
Average putrescibles	19%	100%	19%	31%	5.9%	64% *(50%-80%)*	11.5%	76	210	272
Textiles	39%	50%	20%	2%	0.4%	30%	6%	39	108	503
Miscellaneous combustibles	37%	75%	28%	6%	1.7%	35%	10%	65	179	665
Fines	14%	65%	9%	5%	0.5%	60%	6%	37	103	137
Total					18%					

Table A2.27 Estimates of landfill gas control parameters across the EU

	AU	BE	DK	FI	FR	GE	GR	IR	IT	LU	NL	PO	SP	SW	UK	EU
% of waste in landfill sites with gas control	33%	60%	90%	90%	60%	90%	10%	23%	60%	60%	90%	23%	23%	90%	90%	68%
Methane collection efficiency	20%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	63%	50%	70%	54%
LFG utilisation for energy	20%	60%	60%	60%	60%	60%	60%	60%	60%	60%	60%	60%	73%	60%	60%	60%

Table A2.28: Estimates of CO_2 and methane released from landfilled waste and electricity generated from landfill gas

	CH_4 generated kg/t waste material	CO_2 generated kg/t waste material	CH_4 collected kg/t waste material	CH_4 oxidised kg/t waste material	CH_4 released kg/t waste material	CH_4 used for energy kg/t waste material	Short term CO_2 released kg/t waste material	Electricity generated kWh/t waste material	Avoided emissions kg CO_2/t waste material
Paper	77	212	28	2	49	17	288	71	32
Putrescibles	76	210	28	2	49	17	286	71	32
Textiles	39	108	14	1	25	9	147	36	16
Miscellaneous. combustibles	65	179	24	2	42	14	244	60	27
Fines	37	103	14	1	24	8	140	35	15

Note: Avoided emissions are calculated from EU-average electricity generation emission factor of 0.45 kg CO_2 eq/kWh.

Landfill Gas Control Parameters

Following generation of methane within the landfill site, the amount which actually escapes to the atmosphere is determined by a number of factors. The main parameters are:

- the percentage of the waste which is lying in sites which have gas control,
- the efficiency of gas collection at these sites,
- the proportion of uncollected methane gas which is oxidised in the landfill cover.

We have attempted to collect data on the first two of these parameters for each member state, and where data is missing we have extrapolated either by expert opinion or by matching to other countries with similar waste regimes. We have then derived average figures for the EU, weighted by waste arisings in each country. The figures used are shown in Table A2.27 and are summarised in Table A2.30.

The Landfill Directive requires the implementation of gas control measures on all new landfill sites and almost all existing sites by 2007. There are exceptions only for small existing sites or those serving islands and isolated communities. In the year 2000, we have official estimates only for Austria (33%), Spain (23%) and the UK (90%). We estimate that the proportion of waste in sites with gas control ranges from as little as 10% in Greece to 90% in the UK and Germany. Our estimated weighted average across the EU is 68%.

Estimates vary as to the efficiency with which gas collection systems in landfills gather the methane formed in waste. Gas collection networks installed around the periphery of a site to prevent landfill gas migrating into neighbouring land have very limited efficiency in reducing emissions from across the site. On the other hand, modern, well-designed systems with gas wells installed throughout a landfill site may collect between 70 – 90% of the gas that is formed. We have no data on this parameter so our weighted average is derived from estimates of the likely efficiency in each EU country based on our knowledge of landfill practices in the EU.

We also include an estimate of the proportion of gas which is vented without combustion on sites with gas collection. This represents the gas collected during the first few years of site operation, and during the later years when the peak of methane production is over. During this period the concentration of methane in the gas collected is too low for combustion without the addition of another fuel such as natural gas, and therefore the gas collected is often vented without combustion. Although there is little data on this parameter, we estimate a figure of 10%.

The IPCC guidelines for estimating CH_4 emissions from landfills [43] include a factor to account for methane oxidation. The default value given is zero. Emissions are corrected by a factor of (1-methane oxidation factor). A recent study to estimate emissions of CH_4 from UK landfills adopted a methane oxidation factor of 10%[38]. This value will be used for the present study.

Due to the uncertainty in the estimation of the number of sites with gas control and the gas collection efficiency, we have performed a sensitivity analysis in which we explore three test cases, to illustrate the range of values which may apply:

- **Case 1:** Limited gas collection. Waste is disposed of to a large landfill with peripheral gas collection only (20% of gas is collected). All of the collected gas is flared and there is no energy recovery. The site is capped but imperfections in the capping layer mean that limited methane oxidation takes place (10%).
- **Case 2:** A modern 'best practice' landfill site. Similar to case 1, but with gas collection from the entire site area, for compliance with the Landfill Directive, with an overall collection efficiency of 80%. Most (60%) of the gas collected is used for generating electricity, using a spark-ignition engine as the prime mover. The overall conversion efficiency is 30%. No particular additional measures have been taken to further reduce the escape of un-collected methane, so the methane oxidation rate remains at 10%.
- **Case 3:** Restoration layer approach. A futuristic case, similar to case 2, except that a restoration layer has been added over the top of the cap, which allows escaping CH_4 to come into close contact with methanogens. As a result, the methane oxidation rate for uncollected gas increases to 90%.

Site characteristics are shown in Table A2.29.

Table A2.29: Landfill site characteristics.

	Case 1	**Case 2**	**Case 3**
	Limited collection	Best practice	Restoration layer
Sites with gas collection	100%	100%	100%
Gas collection efficiency	20%	80%	80%
Methane oxidised	10%	10%	90%
Landfill gas used	0%	60%	60%
Power gen efficiency	0%	30%	30%

Carbon sequestration

As described in Section 2.4.1, there is a case for arguing that biogenic organic carbon locked up in landfills can be considered to have been removed from the natural carbon cycle, and therefore should receive a credit for reducing carbon dioxide emissions. The argument is that, for example, plant material which would otherwise have degraded and released its carbon content to the atmosphere will instead be preserved in the anaerobic conditions of a landfill site. This argument can be applied to any biogenic carbon which does not degrade in a landfill site over the time horizon used for the calculation of global warming, which is usually 100 years.

Our estimates of the amount of carbon which is degraded within landfill sites lead automatically to a figure for the amount of carbon sequestered, i.e. not degraded within a 100-year time horizon. This is calculated as shown in Table A2.26, from the difference between DOC and DDOC. It should be noted that all the uncertainties in calculating DDOC lead through directly to equivalent uncertainties in the estimates of carbon sequestered in landfills. The sensitivity ranges tested for the DDOC parameter will also give a sensitivity illustration for carbon sequestration. However, we have also included in the sensitivity analysis a case with zero credit for carbon sequestration.

Avoided energy emissions

After collection, landfill gas can be either flared or used to produce energy (see Box 1). If landfill gas is used to generate energy, there is a credit from avoided fossil fuel emissions. For this study it is assumed that only electricity is generated (no heat). The energy generated depends on the amount of methane formed and collected, which depends on the DDOC of each waste stream as described above, and the collection efficiency

The key parameters are:

- the proportion of collected gas which is burnt for energy recovery,
- and the efficiency of conversion of landfill gas to electricity.

Our estimates for these parameters are shown in Table A2.30. The estimate for the proportion of landfill gas burnt for energy recovery is based largely on a UK estimate of 60%. We had estimates for only 2 other EU countries as shown in Table A2.27, and all other countries were assumed to follow the UK figure.

The calculations of the amount of methane and CO_2 generated and released and the amount of energy generated are shown in Table A2.28.

Table A2.30 : Summary of estimated EU average landfill gas control and recovery parameters

Parameter	Value
Fraction of landfill carbon decaying to methane	50%
Percentage of waste in sites with gas control	68%
Gas collection efficiency	54%
Percentage of uncollected methane oxidised	10%
Percentage of collected LFG vented without combustion	10%
Percentage of collected LFG utilised for energy	60%
Percentage of collected LFG flared	30%
LFG electricity generation efficiency	30%

Box 3: Landfill gas control and energy recovery.

Energy can be recovered from landfill gas by combusting the CH_4, which makes up about 40–65 % by volume. Various uses have been demonstrated for landfill gas:

Electricity generation from landfill gas has proved to be the most popular form of energy recovery, driven in some instances by subsidies and other market incentives. Minimal pre-treatment is all that is usually needed before landfill gas can be used for power generation – for example, the removal of condensed liquids and particles that can damage engines is often sufficient clean-up. The most popular forms of prime mover for power generation are reciprocating engines, mostly based on the spark-ignition design, but some plant also operates on compression-ignition engines. Gas turbines are also deployed. The engines or turbines then drive an alternator to provide electricity to the grid.

Typical unit capacities range from about 250 to 2.5 MW_e for spark ignition engines, 2-10 MW_e for compression ignition engines and >5 MW_e for gas turbines. Examples of steam plant driven by landfill gas and gas turbines operating in combined and open cycle mode are also available, but these are the exceptions. Most European plant is powered by spark-ignition engines built for ship or heavy vehicle propulsion and adapted to run on landfill gas. Overall thermal efficiencies range from about 28-32%. An average of 30% has been adopted for this analysis.

Other uses for energy from landfill gas include CHP operations and as a direct fuel for running nearby industrial processes (brick making being one of the earliest uses for landfill gas in Europe), or as a fuel for landfill vehicles or public transport fleets. Whilst examples of such uses are available, they are exceptions, and so have not been considered in further detail. The great advantage for power generation is that access to the market is relatively straightforward and contracts to take all of the energy produced can usually be negotiated.

OVERALL GREENHOUSE GAS FLUXES FROM LANDFILL

The overall greenhouse gas fluxes from landfill from waste components and from MSW are summarised for the base case assumptions in upper part of Table A2.31. The right-most column shows the total greenhouse gas fluxes. This is calculated as the sum of fossil CO_2 emissions from processing the waste at the landfill, energy use (ie fuel for the compactors etc), avoided energy and materials (in this case avoided emissions associated with electricity generated from landfill gas), transport emissions, sequestered carbon and methane and nitrous oxide emissions. All the data are expressed in kg CO_2 equivalent, per tonne of MSW or MSW component. Data for MSW are calculated by proportioning the amount per tonne of each waste component by the amount of that component in MSW, as shown in Table 9 on page 25.

The results for the base case show that for landfill of MSW, the main contributions to the total greenhouse gas flux of 327 kg CO_2 eq / tonne of waste comes from methane emissions (712 kg CO_2 eq / tonne) and carbon sequestration (-371 CO_2 eq / tonne). A credit of about –22 kg CO_2 eq /tonne comes from avoided emissions due to the use of landfill gas for electricity generation, replacing ‘average’ EU generating plant. The main impacts come from putrescible wastes, paper, fines and miscellaneous combustible (‘other’).

The lower part of the table illustrates the effects of alternative assumptions to the base case. Firstly, if we allow the displaced electricity to come from wind or coal-fired generating plant, it can be seen that the impact on the overall greenhouse gas flux is very small, since the avoided energy term for landfills is itself a small fraction of overall greenhouse gas flux.

On the other hand, the outcome of the analysis is very sensitive to alternative values for DDOC of paper and putrescible wastes. For the low DDOC case, values of 20 and 50 % were chosen for paper and putrescible (compared with 35 and 64 % in the base case). The high DDOC case used values of 50 and 80%. The low DDOC case reduced overall estimates of greenhouse gas fluxes to almost 10% of base case, whilst the high DDOC resulted in an almost two-fold increase.

The table also shows the impact of carbon sequestration on overall greenhouse gas flux. If we leave out the impact of sequestration from the analysis, then the overall flux almost double to just under 700 kg CO_2 eq/tonne.

Finally, the last three lines of the table show the impact of gas collection practices. As expected, the use of limited gas collection increases the total greenhouse gas flux compared with base case. However, ‘best practice’ in gas collection and (especially) the use of a restoration layer to facilitate methane oxidation causes the overall greenhouse gas flux to become negative, taking into account the large negative contribution due to carbon sequestration. Landfills managed to these standards would, according to this analysis, function as a net sink for greenhouse gases.

Table A2.31. Emission factors for landfilled waste (kg CO2 eq/t material treated)

Waste management option	Waste component	Short cycle CO_2	Fossil CO_2				Short cycle C sequestered	Sum of fossil C and sequestered C	CH_4 emission	N_2O emission	Total GHG flux
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	(GWP=-1)		GWP=21	GWP=310	
Landfill											
Landfill gas collected and used for electricity generation. Base case assumptions.	Paper	288	0	1	-32	7	-786	-809	1032	0	**223**
	Putrescible	286	0	1	-32	7	-272	-295	1025	0	**730**
	Plastic	0	0	1	0	7	0	8	0	0	**8**
	Glass	0	0	1	0	7	0	8	0	0	**8**
	Metal	0	0	1	0	7	0	8	0	0	**8**
	Textiles	147	0	1	-16	7	-503	-511	526	0	**15**
	Other	172	0	1	-19	7	-369	-380	617	0	**237**
	MSW	**199**	**0**	**1**	**-22**	**7**	**-371**	**-385**	**712**	**0**	**327**
SENSITVITY ANALYSIS											
Wind power	MSW	199	0	1	0	7	-371	-363	712	0	350
Coal power	MSW	199	0	1	-47	7	-371	-409	712	0	303
Low DDOC	MSW	141	0	1	-16	7	-456	-463	505	0	42
High DDOC	MSW	256	0	1	-28	7	-286	-306	920	0	614
No sequestration	MSW	199	0	1	-22	7	0	-14	712	0	699
Limited gas collection	MSW	184	0	1	0	7	-371	-362	825	0	462
Best practice gas coll	MSW	254	0	1	-48	7	-371	-410	290	0	-121
Restoration layer	MSW	277	0	1	-48	7	-371	-410	111	0	-299

Note that the data are expressed per tonne of material in question. For MSW, the emission factors are estimated from the sum of the constituent waste components multiplied by their relative proportion in the waste stream, as given in Table 9 on page 25. The ‘other’ category refers to the sum of ‘fines’, and miscellaneous combustibles and non-combustibles.

LANDFILL COSTS

Typical gate fees by member state have been taken from three sources: a European Environment Agency report, a database of environmental taxes and direct communications with national experts (see Table A2.32). The sources did not always agree which is why the total gate fee does not always equal the sum of the tax and tax free value. The average fee, weighted by the waste arisings in each country, is 56 Euro/t (including landfill taxes).

Table A2.32 Landfill gate fees and taxes in the EU

Country	Gate fee excl. tax	Source	Tax	Source	Total gate fee
Austria	92	3	21.8-29.1 (43.9 in 2001)	1	117
Belgium	68-83 50	1 3	3.7-22.3	1	63
Denmark	34	3	50	1	84
Finland	12	3	15	1	27
France	32-55	2	9	1	52.5
Germany	26-153 60-90	2 1	(proposed 12.7)	1	90
Greece	10 6-15	3 1	0	1	11
Ireland	44-51	2	0	1	47.5
Italy	10.3-25.8	1	10	1	28
Luxembourg	162	3	0	1	162
Netherlands	75	1	12.4-64.3	1	113
Portugal	6-15	1	0	1	11
Spain	15-30 9 15	1 2 3	0	1	23
Sweden	23-90	2	29	1	86
UK	18-33	1	17.6	1	43

Sources:

1) Database of environmental taxes, http://europa.eu.int/comm/environment/enveco/env_database
2) National expert
3) European Environment Agency 1999, 'Environment in the EU at the turn of the century', chapter 3.7.

Appendix 3. Incineration

INCINERATION PROCESS DESCRIPTION

The purpose of thermal treatment of waste (which in the narrow sense usually means combustion in incinerators) is to reduce the bulk of waste needing ultimate disposal in landfills to an inert inorganic ash residue. Organic carbon compounds are oxidised to CO_2 and water vapour, which are discharged to the atmosphere in the stack gas. Incineration of fossil carbon in plastics (for example) therefore makes a net positive contribution to global warming, but incineration of short-cycle carbon compounds (in paper, food, vegetation etc) is neutral in global warming terms. Residual organic matter remaining in the ash residue should be reduced to a very low level if the combustion process is carried out efficiently. The ash will therefore have virtually no capacity to form organic leachates or gas after disposal in landfills.

Heat, power or both can be recovered from thermal treatment. Most new incinerators are designed for energy recovery, and the proposed incineration directive (98/C/327/07) will require energy to be recovered as far as possible.

Box 4 Development of incineration technologies

Incineration developed during the latter part of the 19th century as a means of reducing the bulk and hazardousness of waste produced in the rapidly growing metropolitan areas. It soon became used as an opportune means of energy recovery. By 1912, there were some 76 incinerators operating in England, recovering energy as heat or electricity. New economic prosperity following the Second World War led to an increase in the amounts of waste produced per head of population, and incineration enjoyed a considerable growth in capacity. However, the communities in which they were located often regarded waste incinerators poorly. Even as late as the 1970s, emission control on incinerators was usually limited to simple cyclones for reducing dust emissions. Poor plant design and operating standards resulted in lack of control over combustion conditions, giving rise to emissions of smoke, odours and high levels of residual organic matter in the ash. Incinerators were identified as major urban sources of heavy metals, dust, acid gases and NO_x, and products of incomplete combustion, such as dioxins and other toxic organic micro-pollutants. Concern over the public health impacts of these emissions led to the introduction of the 1989 incineration directives, the first community wide legislation to set minimum environmental standards for waste incineration. The 1989 directives resulted in the closure of existing plant that could not be upgraded to higher standards, and set minimum limits for all new incinerators. A further tightening of environmental standards for waste incineration will come about through the new incineration directive, which is due to be implemented in 2002.

As a result of strict regulation of incinerators under both community and member state legislation, a very marked improvement in emissions has been achieved. Nevertheless, incineration remains a highly contentious waste management option, not least because of remaining concerns over emissions, especially of dioxins. These issues are not considered further in this study, which is exclusively concerned with the greenhouse gas impacts of waste incineration.

Thermal treatments of waste include combustion-based techniques and advanced thermal conversion (ATC) techniques. The most common form of incineration in use at present is large scale mass burn incineration, with annual throughputs usually in excess of 100,000 t/year. Smaller plants burning specialised wastes or refuse-derived fuels (RDF), sometimes co-fired with peat, wood or coal are also available, often based on fluidised bed combustors. In the future we may see an expansion of ATC options such as pyrolysis/gasification. FBC and pyrolysis reduce overall emissions of harmful combustion products such as nitrogen oxides (NO_x). Waste can also be co-incinerated in power plants, blast furnaces or cement kilns. In this study we assess the following options:

- Mass burn incineration
- Pyrolysis/gasification
- RDF combustion in:
  - Fluidised bed combustors (FBC)
  - Co-combustion in coal-fired power plants and cement kilns
  - Co-combustion in coal-fired cement kilns

These options are described below.

Mass-burn incineration

During combustion, the waste is burnt in the presence of a good supply of air, so that organic carbon is essentially completely oxidised to CO_2, which, along with water vapour and trace products of combustion, is discharged to the atmosphere. Energy is recovered in the form of steam, which is used to drive turbines for electricity generation. Some incinerators may also provide steam or hot water for process or community heating schemes as well as electricity in combined heat and power (CHP) applications. There are two main approaches to waste combustion – mass-burn incineration and process and burn incineration, in which a refuse-derived fuel (RDF) is first prepared.

Mass-burn incineration is currently the most widely deployed thermal treatment option, with about 90% of incinerated waste being processed through such facilities. As the name implies, waste is combusted with little or no sorting or other pre-treatment. Waste arriving at a mass-burn incinerator is tipped into a loading pit and from there transferred by crane and grab system into the combustion chamber loading chute. The waste is then conveyed through the combustion chamber, usually on a moving grate system (of which there are many designs) or through the slow rotation of the combustion chamber itself (rotary kilns). What ever system is used, its purpose is to ensure thorough mixing and even combustion of the waste, so that complete burn-out has occurred by the time the ash residue is discharged into a water-filled quenching tank at the end of the combustion chamber. Air is introduced from below and above the grate at flow rates adjusted to suit the rate of combustion. The hot combustion gases pass through heat exchange sections of the combustion chamber, where steam is generated for energy recovery. The cooling combustion gases then pass through various stages of emission control. These include dry or wet scrubbers for removing acid gases (SO_2, HCl), injection of reducing agents such as ammonia or urea for controlling NO_x emissions, activated carbon injection for dioxin control, and finally particulate removal by filtration or electrostatic precipitators, before the cleaned gases are discharged to the atmosphere.

Mass burn incinerators are specifically designed to cope with all components in the MSW stream, which generally has a relatively low average gross calorific value (GCV), in the range 9-11 GJ/tonne – about one third that of coal or plastics. However, individual types of waste vary markedly in their calorific values, from zero for wet putrescible wastes to over 30 GJ/tonne for some plastics. Loading an even mixture of wastes into the combustion chamber is therefore very important to ensure that the overall heat input stays in 9-11 GJ/tonne range for which the plant is designed to operate. Wastes are therefore mixed in the loading pit to even out obvious differences in composition before loading the combustion chamber. Excess amounts of high CV waste like plastics can lead to high temperature corrosion of heat exchange surfaces due to the high concentrations of chloride found in MSW. The need to avoid high temperature corrosion by limiting combustion chamber temperatures is one of the main reasons why the thermal efficiency of waste incinerators is low, compared with coal-burning steam cycle power stations. On the other hand, if the GCV of incoming waste falls much below about 7 GJ/tonne, then the waste may not burn properly (or even at all) under the conditions inside the combustion chamber, and efficiency of energy recovery would markedly decrease. A pilot fuel would therefore be required to sustain efficient combustion and to ensure that statutory temperature conditions are achieved to prevent the formation of harmful products of incomplete combustion. Such conditions may occur when high quantities of wet garden waste come through the waste stream, especially in spring and autumn.

Energy is recovered from mass-burn incinerators as heat (in the form of steam and hot water). The heat may then be used directly for (eg) district heating, or some of it is converted into electricity, by mean of a steam turbine/alternator. Combined heat and power and heat-only incinerators are widely deployed in some northern European countries, such as Denmark and Germany, where significant markets for the heat exist and where a tradition of investing in the considerable cost of a heat distribution system has grown up. Elsewhere, power only schemes are more widespread, because of the lower costs of marketing electricity, even though, as outlined below, the overall efficiency of power only is much less than that of CHP or heat only. A recent study of energy recovery from waste incineration in a number of western European countries indicated that energy recovery was roughly divided between heat and electricity on the basis of equal amounts of waste going to CHP and power-only plant [48].

Several material streams emerge from mass-burn incineration. The greatest of these is the ash residue discharged from the combustion chamber, which may represent between 20 – 30% of the mass of waste consumed. The ash may be processed by stabilising and grading to form a useful secondary construction material that can be used for low-grade applications such as road or car-park base layers. Re-use of incinerator ash varies from country to country. Half of existing incinerators in the UK and all plants in the Netherlands have an ash processing facility. Ash which cannot be re-used is landfilled. Metals can also be recovered from the bottom ash and sold to reprocessors. In plants with an ash-processing facility, nearly all of the ferrous metal can be recovered, otherwise up to 90% can be recovered. Non-ferrous metal can also be recovered in plants with ash processing.

Emissions standards for incinerators have recently been tightened through new emission limits imposed under the new incineration directive and extensive treatment of the flue gases is necessary to meet the new limits. Residue is produced from the air pollution control system, representing about 2-4% by weight of the incoming waste. This material consists of salts and surplus alkali from acid gas neutralisation, although some plants using wet scrubber systems currently discharge the scrubber residues to water as a salts solution. In addition, fly ash containing dioxin and heavy metals is produced. This material requires disposal at hazardous waste landfills, usually after some form of stabilisation or immobilisation in an inert medium such as cement has taken place. In Germany, salt caverns are used for storage of such hazardous materials.

Mass burn incinerators represent a considerable capital investment, in the order of 75 to 150 million Euros for a medium sized facility of about 400,000 tonnes throughput per year. Capacities range from about 100,000 to over 1 million tonnes of waste per year. Considerable economies of scale apply, especially just above the lower end of this range, and few new mass burn facilities much below this figure exist. Modular designs are common, with larger facilities consisting of several incineration lines working in parallel. The working life of an incinerator is typically around 20-30 years, although extensive maintenance and re-fitting of worn-out parts occurs during the working life.

To be cost-effective, mass burn incinerators require a guaranteed supply of waste within known limits of composition, available throughout the life of the plant. Because of the large scale of operation, such facilities may effectively ‘lock-in’ supplies of waste that could otherwise go for recycling. In addition, the requirement for bulk waste to be provided within a relatively narrow range of calorific value means that removal of particular waste streams for recycling could cause the remaining waste to fall outside the acceptable range. For example, removal of paper and / or plastics for recycling would increase the relative proportion of putrescible waste in the residue and lower its calorific value. On the other hand, removal of putrescible wastes as well, for composting, would help to keep the calorific value of the residue in the acceptable range, but reduce the overall quantity of waste available for processing. Reduction in either the calorific value or quantity of waste consumed would reduce the amount of energy recovered, the sale of which provides one of the main income streams (along with the disposal fee) of the incinerator. Reductions in the sales value of energy would then feed through into higher disposal charges for the waste.

Pyrolysis and gasification

Along with the combustion technologies outlined above, there is increasing interest in the advanced thermal conversion technologies of pyrolysis and gasification as applied to MSW. These technologies differ from combustion in that the waste is first heated in either the complete absence of air, or with a very restricted quantity of air. Organic matter in the waste breaks down thermally to give a mixture of gaseous and/or liquid products that are then used as secondary fuels. The secondary fuels are used to provide the heat input for the process and to run engines for power generation. The only combustion air required is for the engines, so that a very much lower volume of exhaust gas is produced for cleaning than in conventional incineration, allowing much lower emissions to be achieved. Some processes may also produce a solid coke residue that may be used as a coal substitute. A wide range of alternative designs are being developed, but so far there are only about five plants in commercial operation in the EU (all of which are in Germany), with a further eight at the proposal stage. Commercial scales of operation are around 100,000 tonnes/ year of bulk MSW. Similar consideration regarding the demand for available waste of defined composition throughout the life of the plant apply as outlined for mass burn incineration.

Process and burn technologies – refuse derived fuel

Process and burn technologies differ from mass-burn incineration in the amount of pre-treatment applied to the waste, giving rise to a fraction enriched in combustible materials that can be used as a refuse-derived fuel (RDF). Processing the waste allows materials for recycling to be removed from the combustible residue, along with wet organic materials such as food and garden wastes for separate treatment. The combustible fraction (consisting of paper, card, plastic film etc) may then either be burnt directly as a coarse flock (c-RDF) or compressed into dense pellets (d-RDF) for sale as a supplement fuel in industrial boilers. An advantage of the RDF over mass-burn incineration is that because the waste is sorted and shredded before combustion, the combustion equipment can be smaller, less robust and therefore less expensive. Fluidised bed boilers are finding widespread application for RDF combustion, which offers some advantages in terms of ease of emission control. They are also less sensitive to variations in CV of the incoming fuel.

RDF may be used as a fuel source in energy-requiring processes. RDF has, for example, been used with coal, wood and peat for power generation. High heat value wastes are also used in cement manufacture, where they can substitute directly for conventional fuels such as coal. The ash residue in this case becomes incorporated into the cement clinker. It is also possible to separate the plastics fraction of the waste (i.e. the highest calorific value component) for separate incineration, thus avoiding the inefficiencies associated with combustion of lower CV wastes.

RDF and co-incineration are not major disposal routes outside a few niche applications. RDF technology developed considerably in the UK in the early 1980s, but the market has since been static and limited to a few of the surviving initial schemes. Some RDF combustors also operate in Italy, Finland and Germany. Similarly, there have been a few demonstrations of burning RDF with coal in conventional pulverised fuel power stations, but the added complexities of dealing with an additional fuel in the power stations have tended to militate against its more widespread uptake. RDF has been successfully used as a fuel in cement kilns, especially when enriched in high CV plastic waste, but generally the main waste-based fuels used for this purpose have been tyres, solvents and plastics collected from commercial sources.

INCINERATION GREENHOUSE GAS EMISSIONS

The following steps are considered in assessing greenhouse gas fluxes from thermal treatment of MSW:

- **Mobilisation**. This includes collection, sorting and transport of wastes from householders to the thermal processing plant and transport of residues to landfill and recovered ferrous metal to reprocessors.
- **Treatment.** Emissions of fossil-derived CO_2 from the thermal treatment of waste, plus N_2O formed during combustion.
- **Disposal** – all organic carbon is assumed to be destroyed by the thermal treatment and so no greenhouse gas flux comes from the disposal of residues to landfill[a].
- **Displaced emissions** from energy recovered during thermal treatment and recovered metal.

Mobilisation emissions

Waste is collected direct from households and taken either straight to a mass-burn incinerator (in rural areas) or via a refuse transfer station (RTS) in urban areas. The proportion of the population living in urban areas determines the fraction of waste using each route. Bottom ash is transported either to landfill or market depending on the proportion sold (it is currently assumed that all the ash is sold). Fly ash and flue-gas cleaning residues are transported to a hazardous waste landfill site. Recovered metal is taken to a re-processor. Emissions from the RTS are assumed to be negligible in terms of greenhouse gases and have been omitted from further consideration.

---

[a] Strictly speaking, a few percent of the original carbon remains in the bottom ash in an inert form resistant to leaching. This could be considered as sequestered carbon. However, for simplicity, this has been omitted from the model.

Transport routes:
Rural: Household → incinerator
Urban: Household → RTS → incinerator
→market or landfill for bottom ash
→hazardous waste site for fly ash and flue gas cleaning residues
→reprocessor for metal sales

Vehicle emission characteristics and assumed journey distances are given in Appendix 1. Mobilisation emissions are summarised in Table A3.33. The major component of transport emissions (87%) is associated with waste collection and delivery of the MSW to the thermal treatment plant. The remaining emissions come from residue disposal and transport of recovered ash and metals.

Table A3.33: Mobilisation emissions associated with thermal treatment of MSW, residues and recovered metals.

From	To	Load	Vehicle	Per cent of load via journey	CO_2 emissions, kg/tonne of waste
Household	Thermal treatment	MSW	RCV	19%	0.8
Household	RTS	MSW	RCV	81%	3.5
RTS	Thermal treatment	MSW	L1	81%	2.9
Thermal treatment	hazardous landfill	Ash and FG residues	L1	100%	0.4
Thermal treatment	market	Ash	L2	100%	0.2
Thermal treatment	Landfill	Ash	L1	0%	0.0
Thermal treatment	Metals reprocessor	Metal scrap	L2	100%	0.3
Total					**7.7**

Mass burn incineration is the dominant thermal treatment considered in this study. However, some analysis has also been undertaken on RDF combustion, for which the mobilisation emissions are expected to be closely similar to the mass-burn situation. Given the low overall greenhouse gas flux associated with mobilisation, this step has not been explicitly modelled for RDF, and the values used for mass burn incineration have been employed instead.

Treatment emissions

The carbon present in waste is converted to carbon dioxide during incineration. For biogenic materials, this is short-term carbon dioxide whereas for plastics and synthetic materials it is derived from fossil fuels. Emissions are calculated from the figures for total carbon content in the waste, and assumptions regarding the proportion of carbon which is short-term or fossil-derived for each waste stream. Emissions of nitrous oxide during combustion were also estimated. The estimates are shown in Table A3.35.

Carbon contents were derived mainly from the UK analysis of household waste [44]. However, the carbon content of plastic waste derived from this study (52%) seemed low in comparison with other estimates and was therefore adjusted on the basis of more recent figures shown in Table A3.34. The composition of EU plastic waste (the mix of resins) was taken from APME data [49]. The carbon content of pure resins was worked out based on their chemical formulae, and the net calorific value of the resins were taken from [50]. The carbon content and calorific value of wet waste was then worked out assuming a typical 10% water content.

Table A3.34 Carbon content and calorific value of plastic waste. See references 49 & 50.

		Pure resin		Wet waste	
Plastic	**% of EU plastic waste**	**carbon content**	**Net CV MJ/kg**	**carbon content**	**Net CV MJ/kg**
LDPE[a]	21%	86%	45	70%	38.68
HDPE	18%	86%	45	70%	38.68
PP	20%	86%	46	70%	38.68
PVC	9%	38%	18	39%	18.08
PET	9%	63%	22	55%	22.7
PS	11%	92%	41	70%	34.75
PU	3%	58%	25	52%	22.26
Others	9%	75%	35	61%	31
Weighted average	**100%**	**78%**	**35**	**61%**	**31**

N_2O from incinerators

Emissions of nitrous oxide from incinerators have also been estimated. Data is sparse because nitrous oxide emissions from incinerators are not regulated and hence very few published measurements are available. For mass burn incinerators, data has been taken from two sources: range estimates from the UK and Germany quoted in the IPCC Good Practice guidelines [51] and an estimate for an incinerator in Paris [9]. We have taken an average of the mid-points of these three values.

The IPCC Good Practice guidelines suggest that emissions may be higher for fluidised bed combustion than for mass burn combustion. The only figures quoted are for a Japanese wet process, where emissions from FBC are 4 to 6 times higher than for mass burn combustion for MSW, and 1 to 3 times higher for sewage sludge combustion. However, the wet process is not applicable in the EU situation, and earlier figures from the revised 1996 guidelines do not show any significant difference between mass burn and FBC. In the absence of reliable data we have assumed that emissions from fluidised bed combustion are twice as high as those for mass burn combustion. For pyrolysis we assume that emissions will be lower than for mass burn due to the smaller volumes of flue gas involved. In the absence of any data we assume zero emissions from pyrolysis.

---

[a] LDPE=low density polyethylene, HDPE=high density polyethylene, PP=polypropylene; PVC=polyvinyl chloride; PET=polyethyleneterephthalate; PS=polystyrene, and PU=polyurethane.

Table A3.35: Incineration treatment emissions

Component	Carbon content %C	% fossil carbon	fossil CO_2 kg/t	N_2O kg/t
	a	b	c=a*b*44000/12	
Paper/Card	33%	0%	0	0.05
Putrescible	19%	0%	0	0.05
Plastic	61%	100%	2237	0.05
Glass*	0	0%	0	0.05
Metals	0	100%	0	0.05
Textiles	39%	50%	718	0.05
Other**	24%	29%	256	0.05

**The chemical analysis of dustbin waste shows a small amount of carbon associated with glass and metal as food remnants and paper labels on cans and bottles, but this has been omitted for simplicity

**The 'Other' category is a weighted average of the three UK categories of 'fines', 'miscellaneous combustibles' and 'miscellaneous non-

Energy use

The energy used for mass burn incineration of bulk MSW (for operation of the crane, grate, air pollution control system, fans, cooling water pumps, ash handling etc) is assumed to be provided by in-house generated electricity. The internal use of energy is reflected in the overall thermal efficiency of the plant, described in the following section. No allowance has been made for this in-house use for incinerators that do not recover energy.

Fuel preparation for RDF combustion in FBC incinerators and pyrolysis / gasification is limited to segregation and separation from recyclable and non-combustible materials. Energy for this comes from in-house generation. This is taken into account in the overall efficiency of the plant.

For RDF pellet combustion in power plants and cement kilns, we need to take account of the energy used for shredding and compacting the waste into RDF. For this we have taken figures from an APME report [53] which described the energy used to separate a plastics and paper fraction from household waste and process it into fuel pellets (1.76 GJ/t waste, of which approximately two thirds is gas and the remainder EU average electricity). The pellets then need to be pulverised and injected into the plant. Here we must add the energy used to pulverise and inject the fuel, minus the energy needed to pulverise, dry and inject an equivalent (in energy terms) amount of coal. This equates to 0.31 GJ/t waste, which we assume is provided EU average electricity plant mix. The overall emission factor for RDF preparation and injection for combustion in power plants and cement kilns is 111 kg CO_2 eq/tonne of waste[a].

Displaced emissions

Displaced energy

Displaced emissions are those associated with the power and/or heat recovered by the thermal process that would otherwise have to be generated by other processes. As discussed previously, the avoided emissions will depend on the actual power and heat sources displaced, and this will vary from location to location. For the base case, we have assumed that power and heat replaced are those of average EU power and heat generation. However we have examined the

---

[a] This value may overestimate the energy use for cement making since the pellets may not always be shredded in this application, or indeed only coarsely-shredded waste, rather than pellets, may alternatively be used.

sensitivity of this assumption by also testing the cases where coal steam cycle plant or wind power are replaced. A few existing incinerators do not recover energy at all, in which case the displaced emissions will obviously be zero.

Displaced emissions depend on the calorific value of the waste, the efficiency of heat and power recovery at the combustion plant and the emission factor for greenhouse gases that would have been released by the generation of the same amount of energy from the replaced source. Waste incinerators tend to treat either whole MSW (ie in mass-burn applications) or else a RDF prepared from high CV wastes separated out from the bulk MSW.

Table A3.36 shows the calorific values used in this study for the various waste components. The overall CV of MSW is estimated by summing the product of the individual waste components and their proportion in the bulk MSW. For estimating displaced emissions, we need to use the net CV, rather than gross CV, as explained in Box 5. Net CV (NCV) is a measure of the overall useful energy recovered from the waste, taking account of heat needed to dry the waste, and the heat lost to the atmosphere in hot gases and non-condensed water vapour, and in the hot ashes discharged from the furnace.

Table A3.36 also shows the CV of RDF, expressed in terms of the quantity of the whole MSW from which the RDF is prepared, assuming that all of the paper/card, plastics and textiles are diverted to RDF manufacture. The effect of segregation on the CV of the RDF as burnt, expressed in terms of CV per tonne of *RDF*, is shown in Table A3.37. It can be seen from comparison with the results shown in the penultimate line of Table A3.36 that the RDF fraction is significantly enhanced in terms of CV.

To complete the calculation of displaced greenhouse gas emissions, we also need estimates of the thermal efficiency of the waste combustion plant and the emission factors for the displaced energy source. These results are shown in Table A3.38.

The overall thermal efficiency of incinerators depends on the proportion of useful heat that can be recovered from the burning fuel and the amount of energy recovered that is used for on-site services, such as waste and residue handling, air pollution control etc. Thermal efficiencies for power generation range from about 15 to 22% in thermal treatment plant, and for heat a value of about 50% would be typical. However, plants that evaporate liquid effluents from wet scrubbers to avoid producing a liquid effluent would have efficiencies towards the lower end of this range. Even lower efficiencies would result if ash vitrification (melting) was to be implemented, but this is currently uncommon in European plant. Typical thermal efficiencies of 18 and 50% for power and heat recovery using mass burn technology appear reasonable. These efficiencies take into account the own-use of energy within the plant. For MSW of the composition assumed in this study, these efficiencies correspond with electricity and heat outputs of 427 and 1185 kWh/tonne (Table A3.38). In comparison, Reimann quotes typical electricity outputs from MSW incineration in the range 300 to 700 kWh/tonne, with ~1250 kWh/tonne for heat output to district heating [52]. The efficiency of FBC plant burning RDF is similar to that for mass burn plant. For co-incineration of RDF in coal-fired power plants, trials have shown that co-combustion of plastics waste does not affect the overall efficiency of the power station [53]. Therefore we have assumed a typical pulverised coal fired steam cycle power plant efficiency of 36% for co-combustion of RDF. By the year 2020, we assume that efficiencies for all plant will have improved by a few %, as shown in the table, but this is conditional on there being no further increase in emission standards that would require the use of more in-house energy use for emission control. This is considered reasonable, given that the latest incineration directive will only come in 2002.

For co-incineration of RDF in cement making, the waste is assumed to substitute for the replaced fuel (coal) on a heat equivalent basis (i.e. one GJ of RDF substitutes 1 GJ of coal). The carbon content of coal is taken to be 25.8 kg C/GJ.

Table A3.36: Calorific values for MSW and MSW components

Component	RDF components	Per cent by weight in raw MSW	Calorific value of individual waste components, GJ/tonne of waste component		Calorific value contributed by waste components, GJ/tonne MSW	
			GCV	NCV	GCV	NCV
Paper/card	✓	29	13.13	11.5	3.8	3.3
Putrescibles		31	5.9	3.98	1.9	1.3
Plastic	✓	8	33.5	31.5	2.7	2.6
Glass*		11	0	0	0.0	0.0
Metals*		5	0	0	0.0	0.0
Textiles	✓	2	16.11	14.6	0.3	0.3
Othe**r		13	10	8.4	1.3	1.1
MSW		100			**10.0**	**8.5**
MSW as RDF					**6.8**	**6.2**

GCV and NCV are the gross and net calorific values, respectively.

**The chemical analysis of dustbin waste shows a small calorific value associated with glass and metal as food remnants and paper labels on cans and bottles, but this small value has been rounded to zero.

**The 'Other' category is a weighted average of the three UK categories of 'fines', 'miscellaneous combustibles' and 'miscellaneous non-

Table A3.37: Calorific value of RDF

Component	Per cent by weight in RDF	Calorific value of individual waste components, GJ/tonne of waste component		Calorific value contributed by waste components, GJ/tonne RDF	
		GCV	NCV	GCV	NCV
Paper/card	74	13.13	11.5	9.8	8.5
Plastic	21	33.5	31.5	7.0	6.6
Textiles	5	16.11	14.6	0.8	0.7
RDF	**100**			**17.6**	**15.9**

Table A3.38: Displaced emissions from waste incineration, kg CO2 eq/tonne of MSW.

Option	Fuel	Year	Energy form	Energy recovery, %	Energy recovered	Units	Electricity emission factors				Heat emission factors	
							Average EU 2000	Average EU 2020	Coal	Wind	Coal in cement kilns	Average EU heat
							Emission factors					
							kg CO_2 / kWh				kg CO_2 / GJ	kg CO_2 / kWh
							0.45	**0.40**	**0.95**	**0.009**	**94.6**	**0.28**
Mass burn incineration	MSW	2000	Electricity	18%	427	kWh/tonne MSW	190	171	405	4		
	MSW	2020	Electricity	21%	498	kWh/tonne MSW	222	200	473	4		
	MSW	2000 & 2020	Heat	50%	1185	kWh/tonne MSW	529	476	1126	11		338
Pyrolysis & gasification	MSW	2000	Electricity	18%	427	kWh/tonne MSW	190	171	405	4		
	MSW	2020	Electricity	21%	498	kWh/tonne MSW	222	200	473	4		
FBC	RDF	2000	Electricity	18%	309	kWh/tonne MSW	138	124	293	3		
	RDF	2020	Electricity	21%	360	kWh/tonne MSW	161	145	342	3		
Co-incin in coal-fired power stations	RDF	2000	Electricity	36%	617	kWh/tonne MSW	275	248	587	6		
	RDF	2020	Electricity	39%	669	kWh/tonne MSW	298	268	635	6		
Co-incin in cement kilns	RDF	2000 & 2020	Heat	100%	6.2	GJ/tonne MSW					584	

Recovery of ash and metals

Emissions savings arise from any materials recovered from incineration: ferrous metal (principally steel sheeting recovered from tin-plated cans), aluminium and ash used as a secondary aggregate.

We assume that 90% of the steel input to incinerators can be recovered from the ash [107]. Production of steel from recycled materials saves some 1,810 kg CO_2 / tonne of metal, compared with virgin materials (as described in Appendix 7). This figure is for source-separated (non-detinned) scrap. The steel recovered from incinerators fetches a lower price than source-separated scrap but there should be no reductions in recycling yield, so emissions savings per tonne of metal recovered should be approximately the same. In fact emissions savings may be slightly greater as the de-tinning stage is avoided (tin migrates into the steel during incineration, making de-tinning impossible, but modern recycling processes can cope with the extra tin). However the de-tinning stage only accounts for around 10% of the energy budget so this has been neglected.

Box 5 Net and Gross Calorific Values

In assessing the energy available for recovery from wastes by incineration it is necessary to distinguish between the gross and net calorific value of the waste.

The gross calorific value (GCV) is the amount of heat released when all of the combustible material is burnt, converting all of the carbon to CO_2 and all of the hydrogen to H_2O. (It also includes the energy released by the oxidation of other elements such as sulphur and nitrogen, but the contribution of these may usually be ignored for practical purposes since their concentration in wastes is usually very low). Water is assumed to be recovered as liquid and, along with any ash residue, is assumed to have a final temperature the same as the starting material. The GCV may therefore be considered to represent the theoretical maximum amount of energy available through combustion.

In practice, incinerators cannot recover all of the heat implied by the GCV of the waste. This is because the water produced by the oxidation of hydrogen in the fuel is not condensed, but escapes from the system in the stack gas as steam (at about 200-250 deg C) and other residues and products leave the incinerator at a higher temperature than they enter, so removing heat. The energy that would therefore have been recovered by condensing the steam to liquid water is therefore lost to the system. Furthermore, non-combustible residues (ash etc) also remove heat from the incinerator, proportional to their specific heat and the temperature difference between the incoming waste and the discharged hot ash as it leaves the incinerator. Finally water present in the waste will consume energy through evaporation and so reduce the overall amount of useful heat that can be recovered. Thus the wetter the fuel and the more ash it produces, the lower will be the hear recovered. Some waste may therefore remove more heat from an incinerator than they provide through combustion. The more useful parameter for estimating the energy input to incinerators is therefore the net calorific value, which takes account of these potential losses.

The values for calorific value shown in Table A3.36 are average Net calorific values, based on analysis of extensive samples of waste received during the National Household Waste Analysis Programme. Some variation is seen in NCV within categories of waste, reflecting its natural variability. For example, food and garden wastes will contain tree prunings and dry stale bread with relatively high NCV values (ca 10 MJ/kg), to wet grass cutting negative value for NCV may occur for very wet wastes where more heat is removed in evaporating the water in the incinerator than is provided by combustion of the residue. Some variation is therefore to be expected, but overall averages are in good agreement with data reported elsewhere (eg US data reported in [54]) and the overall gross CV of waste based on this composition is within the range of 9 to 11 MJ/kg reported for European MSW.

For non-ferrous metal (i.e. aluminium) the recovered metal may be of slightly lower quality due to partial oxidation in the incinerator. Recovery of non-ferrous metal from incinerator residues is not currently widespread and so has been omitted from the analysis. Similarly impacts due to the use of incinerator ash as a secondary aggregate has not been taken into account due in part to the complexity of deciding what type of material (primary or other secondary aggregate) it replaces. This omission is not thought to make a significant difference to the analysis.

OVERALL GREENHOUSE GAS FLUXES FROM INCINERATION

Summary results of the greenhouse gas fluxes from thermal treatments of MSW and its components are shown in Table A3.39, which recapitulates some of the information presented earlier in this Appendix. It is important to remember that the information in the table shows the contribution of the various waste components incinerated *as part of* MSW or in RDF – it should not be taken to imply that individual waste streams would necessarily be incinerated alone. An obvious example here is putrescible waste, which although present as part of mixed MSW would not be suitable for incineration by itself.

The column showing avoided energy and materials is includes greenhouse gas emissions avoided by recovering ferrous metal from combustion residues (for non-RDF applications) and replaced energy that would have come from other sources. The base case replaced energy is EU-2000 average electricity and heat (with emission factors given in Figure 5 and Figure 6). Alternative sources for electricity replaced have been considered for energy-recovering options, except where coal is, by definition, being replaced by RDF in power stations or cement kilns. The alternative energy sources considered are wind and coal power sources. These are shown for MSW only (not the separate components) in the shaded rows. Note that the data for the RDF options is based on the fluxes from a unit of MSW that goes to make the RDF, rather than the mass of the RDF itself. Total greenhouse gas fluxes associated with RDF manufacture may therefore be compared directly with MSW incineration by summing the fluxes due to RDF and those due to the treatment of the residual material from RDF manufacture (eg landfill, MBT, recycling etc), as presented in the Results section of the main report.

Examination of the table shows the pivotal effects of replaced energy source on the outcome of the analysis for total greenhouse gas flux. For example, if we take the case of mass burn incineration with energy recovery as electricity only (bottom of first part of the table), then the base-case indicates a total greenhouse gas flux due to this option of –10 kg CO_2 /tonne MSW (see right-most column of the table). If the replaced energy came from a wind farm (as shown by the next line) the small net saving in emission increases to a net *positive flux* of 177 kg CO_2 /tonne MSW. On the other hand, if the replaced power were to have come from a coal-fired power station, then incinerator would return a net *reduction* in emissions, of –255 kg CO_2 /tonne MSW. Note that in the case of CHP, the replaced energy relates only to the electricity component – the heat source is not varied in this analysis.

Table A3.39: Emission factors for incinerated wastes (kg CO2 eq/t material or MSW treated).

Avoided energy is assumed to come from the average of EU electricity and heat for year 2000, except for the cases where RDF is incinerated in coal fired power stations or cement kilns, in which case coal is displaced on an energy equivalent basis. Sensitivity analysis is conducted for wind and coal fired electricity where appropriate.

Waste management option	**Waste component**	**Short cycle CO_2**	**Fossil CO_2**				**Short cycle C sequestered**	**Sum of fossil C and sequestered C**	**CH_4 emission**	**N_2O emission**	**Total GHG flux**
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	**(GWP=-1)**		GWP=21	GWP=310	
Mass Incineration Burn no energy recovery. Fe metal recovered from bottom ash.	Paper	**1209**	0	0	0	8	0	**8**	0	15	**23**
	Putrescible	**692**	0	0	0	8	0	**8**	0	15	**23**
	Plastic	**0**	2237	0	0	8	0	**2244**	0	15	**2259**
	Glass	**0**	0	0	0	8	0	**8**	0	15	**23**
	Metal	**0**	0	0	-1369	8	0	**-1361**	0	15	**-1346**
	Textiles	**718**	718	0	0	8	0	**726**	0	15	**741**
	Other	**538**	256	0	0	8	0	**264**	0	15	**279**
	MSW	**652**	230	0	-72	8	0	**166**	0	15	**181**
Mass burn incineration energy recovered as electricity only. Fe metal recovered from bottom ash.	Paper	**1209**	0	0	-257	8	0	**-250**	0	15	**-235**
	Putrescible	**692**	0	0	-89	8	0	**-81**	0	15	**-66**
	Plastic	**0**	2237	0	-703	8	0	**1541**	0	15	**1556**
	Glass	**0**	0	0	0	8	0	**8**	0	15	**23**
	Metal	**0**	0	0	-1369	8	0	**-1361**	0	15	**-1346**
	Textiles	**718**	718	0	-326	8	0	**401**	0	15	**415**
	Other	**538**	256	0	-187	8	0	**77**	0	15	**91**
	MSW	**652**	230	0	-262	8	0	**-25**	**0**	**15**	**-10**
Wind power	MSW	**652**	230	0	-76	8	0	**162**	**0**	**15**	**177**
Coal power	MSW	**652**	230	0	-478	8	0	**-240**	**0**	**15**	**-225**

Note that the data are expressed per tonne of material in question. For MSW, the emission factors are estimated from the sum of the constituent waste components multiplied by their relative proportion in the waste stream, as given in Table 9 on page 25. The ‘other’ category refers to the sum of ‘fines’, and miscellaneous combustibles and non-combustibles.

Table A3.39 continued

Waste management option	**Waste component**	**Short cycle CO_2**	**Fossil CO_2**				**Short cycle C sequestered**	**Sum of fossil C and sequestered C**	**CH_4 emission**	**N_2O emission**	**Total GHG flux**
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	**(GWP=-1)**		GWP=21	GWP=310	
Mass Burn incineration with CHP. Fe recovered from bottom ash.	Paper	**1209**	0	0	-714	8	0	**-706**	0	15	**-691**
	Putrescible	**692**	0	0	-246	8	0	**-239**	0	15	**-224**
	Plastic	**0**	2237	0	-1950	8	0	**295**	0	15	**310**
	Glass	**0**	0	0	0	8	0	**8**	0	15	**23**
	Metal	**0**	0	0	-1369	8	0	**-1361**	0	15	**-1346**
	Textiles	**718**	718	0	-903	8	0	**-177**	0	15	**-162**
	Other	**538**	256	0	-520	8	0	**-256**	0	15	**-241**
	MSW	**652**	230	0	-601	8	0	**-363**	**0**	**15**	**-348**
Wind power	MSW	**652**	230	0	-414	8	0	**-176**	**0**	**15**	**-161**
Coal power	MSW	**652**	230	0	-816	8	0	**-578**	**0**	**15**	**-563**
Gasification / pyrolysis of MSW, energy recovered as electricity only. Fe metal recovered	Paper	**1209**	0	0	-229	8	0	**-221**	0	0	**-221**
	Putrescible	**692**	0	0	-79	8	0	**-71**	0	0	**-71**
	Plastic	**0**	2237	0	-625	8	0	**1620**	0	0	**1620**
	Glass	**0**	0	0	0	8	0	**8**	0	0	**8**
	Metal	**0**	0	0	-1369	8	0	**-1361**	0	0	**-1361**
	Textiles	**718**	718	0	-289	8	0	**437**	0	0	**437**
	Other	**538**	256	0	-167	8	0	**97**	0	0	**97**
	MSW	**652**	230	0	-241	8	0	**-3**	**0**	**0**	**-3**
Wind power	MSW	**652**	230	0	-75	8	0	**163**	**0**	**0**	**163**
Coal power	MSW	**652**	230	0	-433	8	0	**-195**	**0**	**0**	**-195**

Table A3.39 continued

Waste management option	**Waste component**	**Short cycle CO_2**	**Fossil CO_2**				**Short cycle C sequestered**	**Sum of fossil C and sequestered C**	**CH_4 emission**	**N_2O emission**	**Total GHG flux**
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	**(GWP=-1)**		GWP=21	GWP=310	
RDF combustion in FBC. Energy recovered as electricity only	Paper	**1209**	0	0	-257	8	0	**-250**	0	30	**-220**
	Plastic	**0**	2237	0	-703	8	0	**1541**	0	30	**1571**
	Textiles	**718**	718	0	-326	8	0	**401**	0	30	**431**
	MSW	**363**	196	0	-138	3	0	**61**	**0**	**12**	**73**
Wind power	MSW	**363**	196	0	-3	3	0	**197**	**0**	**12**	**208**
Coal power	MSW	**363**	196	0	-294	3	0	**-94**	**0**	**12**	**-83**
RDF combustion is coal-fired power plant	Paper	**1209**	0	111	-1092	8	0	**-973**	0	15	**-958**
	Plastic	**0**	2237	111	-2982	8	0	**-626**	0	15	**-611**
	Textiles	**718**	718	111	-1382	8	0	**-544**	0	15	**-529**
	RDF	**363**	196	43	-585	3	0	**-342**	**0**	**6**	**-337**
RDF replaces coal in cement kiln	Paper	**1209**	0	111	-1092	8	0	**-973**	0	15	**-958**
	Plastic	**0**	2237	111	-2982	8	0	**-626**	0	15	**-611**
	Textiles	**718**	718	111	-1382	8	0	**-544**	0	15	**-529**
	MSW	**363**	196	43	-585	3	0	**-342**	**0**	**6**	**-337**

INCINERATION COSTS

Typical gate fees by member state have been taken from three sources: European Environment Agency report, a database of environmental taxes and direct communications with national experts (see Table A2.32). The sources did not always agree which is why the total does not always equal the sum of tax and tax-free gate fee. The average fee, weighted by the waste arisings in each country, is 64 Euro/t (including taxes) with energy recovery and 66Euro/t without energy recovery . No data were found for Ireland and Greece, where there is currently no waste incineration capacity, and Portugal.

Table A3.40 Incineration gate fees and taxes in the EU

Country	Gate fee excl. tax	Source	Tax	Source	Total gate fee
Austria	105	3	14-71	3	148
Belgium	70	3	3.7-22.3	1	83
Denmark	42-75	3	38 with energy recovery else 44	1	97 or 103 (no energy recovery)
Finland	52	3			52
France	88	3			88
Germany	88	3			88
Luxembourg	132				132
Netherlands	84	3	0	1	84
Spain	32	2			32
Sweden	35	3	0	1	31
	17-45	2			
UK	50	3			50

Sources:
1) Database of environmental taxes, http://europa.eu.int/comm/environment/enveco/env_database
2) National expert
3) European Environment Agency 1999, ‘Environment in the EU at the turn of the century’, chapter 3.7.

Appendix 4. Mechanical Biological treatment (MBT)

MBT PROCESS DESCRIPTION

Mechanical-biological treatment involves the mechanical sorting of whole waste into a biodegradable fraction and a reject fraction. The biodegradable fraction can then be composted (or anaerobically digested) prior to landfill to reduce methane generation.

The system generally operates by sorting the waste prior to composting to remove the non-biodegradable components, typically in a homogenisation drum where the waste is tumbled in a rotating drum for periods of several hours to several days. The degradation is assisted by the addition of water. The material is then screened to remove the materials that have not broken down. These are principally textiles, plastics and metals. Obviously there are some organic materials mixed with these rejects but the proportion is small. Metals are removed for recycling, and the remainder of the ‘reject fraction’ is either landfilled or used as landfill cover/restoration material or incinerated.

The ‘biodegradable fraction’ which passed through the screen is either composted or anaerobically digested. The composting methods used are similar to those described in Appendix 5. Biofilters may be used to reduce odours and bioaerosols, and in some cases additional filters are used to control release of volatile organic compounds.

The composted residue from the MBT process is volume-reduced and has a much reduced capacity to produce landfill gas and leachate after disposal in landfills than untreated waste. Being made from non-source segregated waste, the product is not of sufficient quality to be suitable for agricultural or horticultural use, although it is suitable for use as landfill cover material or can be used in an application such as land restoration at the landfill. In Austria, the use of MBT residue as soil improver is forbidden. The MBT process can therefore be seen as an alternative or complementary pre-treatment for bulk MSW prior to disposal of the residue in landfills.

Various processes are integrated into MBT and a wide range of plant configurations exist. The MBT facility is usually located on an operating landfill to minimise onward transport of the treated waste. An example of a process stream is as follows. Residual MSW is delivered to the site and crushed. The crushed waste is then fed to a screening drum. Material <100-150 mm is passed to a composting drum, after magnetic removal of ferrous metal for recycling. The material in the composting drum is agitated by periodically rotating the drum, and water is added to maintain moisture levels, and sometimes sewage sludge is added for co-composting. Oversize material is kept out of the composting drum and segregated for incineration as RDF or direct landfilling. The composted waste may then be sent for maturation in compost piles, or subjected to an optional period of anaerobic digestion prior to final maturation. The maturation composting step may take place in vessels or in aerated static piles or in windrows turned and watered every few days mechanically. Alternative systems have also been demonstrated in which the entire process takes place in static aerated windrows [55].

The extent to which labile organic matter is degraded during MBT depends to a large extent on the duration of the composting stage and any following period of maturation. Some German systems which achieve a very high degree of stabilisation (as measured by respirometric tests on the compost) may take several months, while the lower levels of stabilisation required under Austrian and Italian standards may be achieved within 40 to 60 days [56]. The process is usually extensively monitored and controlled by computerised systems to ensure reliable and reproducible performance.

Box 6 The use of MBT in Europe

MBT is currently deployed mostly in Germany and Austria. In Germany, the 'Kreislaufwirtschaftgesetz' (closed cycle economy and waste act) which governs waste management gives priority to waste avoidance. Residual wastes that cannot be recovered or recycled have to be pre-treated before landfilling to avoid risks from gas, leachate and landfill settlement. Until recently, only pre-treated waste with total volatile solid content of less than 5% could be landfilled, under the technical standards set in Technische Anleitung Siedlungsabfall (TASi) [57]. As a result of this, only thermal pre-treatment of waste was legally acceptable. However, this has recently been repealed and waste with a total volatile solid content of up to 16% may now be landfilled, paving the way for the use of MBT[58]. MBT had previously been accepted on an experimental basis and in 1999 there were some 20 plants operating, treating one million tonnes of waste, including some industrial scale plants [59]. The German Federal Research Ministry has launched a research programme to evaluate MBT. The results indicate that MBT is a suitable pre-treatment for waste prior to landfill and considerably reduces the environmental impacts of landfilling. The preliminary results from the programme have been reported by Soyez et al [59].

In Austria, MBT is an accepted method of pre-treating waste before landfill. Non-pre treated waste will no longer be permitted in Austrian landfills after 2004. According to the Austrian Landfill Regulation (164/1996), both incineration and MBT are accepted as long as the residue achieves certain standards. For MBT, the main criterion is that the waste has a gross calorific value of less than 6 MJ / kg TS (total solid). According to Raninger et al (1999) [60], approximately 50% of residential waste will be treated by MBT. This will be achieved through the use of nine existing plants with a combined capacity of 300 ktonne/year, plus additional standby sites for upgrading and new facilities designed to meet the acceptance criteria.

The long-term behaviour of highly stabilised MBT residue has been predicted from a series of detailed experiments using landfill simulation reactors [61]. The results showed that:

- MBT reduces the landfill gas emission potential by 90% compared with untreated MSW. The remaining emission potential is characterised by half-lives of 15 – 30 years, about 10 time longer than for untreated MSW. The authors conclude that the slow rate of residual CH_4 emission means that methane oxidising organisms in the cover soil will, in all probability, oxidise all of the CH_4 released.

- MBT residual waste can be compacted to very high density in landfills (ca 1.5 tonnes / m^3, which results in very low hydraulic conductivities (in the range 1×10^{-10} to 5×10^{-9} m/s). As a consequence of the low infiltration of water, leachate production is minimised and the total nitrogen and total carbon content of the leachate reduced by up to 95% and 80 - 90 % respectively.

Field experiments in landfilling MBT residues have shown several benefits in the way landfills can be operated [62]. The MBT residue can be compacted to very high density (~1.5 t/m^3) using conventional landfill compactors. The working area of the site may be kept to a minimum, to reduce the opportunities for water to get into the waste, contouring and covering the filled area as appropriate. Settlement is minimal because of the low residual level of biological activity. The finished area should be covered first with a permeable drainage layer and then with a soil –MBT compost mix of about 0.8 m depth to act as a methane oxidation layer. The trials showed that anaerobic conditions develop in the compacted MBT residue, but because waste is very resistant to further anaerobic decay very little gas is formed and that which is formed is completely oxidised. The need for further gas collection is therefore avoided. The high compaction density and very low infiltration rate of water also reduces the amount of leachate requiring treatment.

MBT GREENHOUSE GAS EMISSIONS

The following steps are considered in the analysis:

- **Mobilisation.** This includes collection and transport of waste from the householders to the MBT plant, transport of RDF to the incinerator and recovered metal to the reprocessor, and transport of incinerator residues. The MBT plant is assumed to be located at a landfill, incurring no significant transportation of the compost product.
- **Treatment.** Emissions associated with the MBT process, landfilling the residue and incinerating the RDF fraction.
- **Disposal.** Greenhouse gas emissions and carbon sequestered in the landfill.
- **Displaced emissions.** Avoided heat and power generation emissions from incinerating the RDF plus any energy avoided through landfill gas collection and use.

The model assumes that after separation of metals, the material is separated into a compostable fraction and a reject fraction. The compostable fraction can be used as input to an AD plant, but we assume that the composting route is followed, with disposal of the composted residue to landfill. For the reject fraction we model two routes: landfill or incineration. At present the relatively high cost of incineration means that landfill is the more usual route, but the incineration option may be more significant in future scenarios. For example in Austria all MBT rejects are currently landfilled but by 2010 it is expected that they will all be incinerated [64].

We assume that the compostable fraction contains 80% of the paper, all of the fines and putrescible material, and 50% of the miscellaneous combustibles. The division of the input waste stream between treatment routes is shown in Table A4.41.

Table A4.41 Modelling of MBT waste stream

Component	% of waste stream	Recycled	Composting, followed by landfilling	Diverted to landfill or incineration
Paper	29%		80%	20%
Putrescibles	31%		100%	
Plastics	8%			100%
Glass	11%		100%	
Metals	5%	100%		
Textiles	2%			100%
Fines	5%		100%	
Misc comb	6%		50%	50%
Misc non comb	2%			100%
Fraction of input waste stream		5%	74%	20%

Mobilisation emissions

Emissions associated with transporting MSW to the MBT plant (which is assumed to be located on a landfill that receives the composted waste), the reject fraction to thermal treatment, and recovered metals to market are summarised in Table A4.42. Vehicle emission characteristics and assumed journey distances are given in Appendix 1.

Transport routes:

Household → MBT plant → compost and reject fraction to landfill
→ recovered metal to reprocessor
→ combustibles to incinerator

The MBT plant is assumed to be situated at a landfill, so there are negligible transport implications of transporting the landfilled residues.

The model indicates that for a tonne of waste of average EU composition, 0.05 tonnes of metals would be removed, 0.30 tonnes composted residue would be produced (assuming a compost yield of 0.4 tonnes per tonne of waste), and 0.20 tonnes reject material would be landfilled or incinerated.

Table A4.42 Mobilisation emissions for MBT

From	To	Load	Vehicle	% by route	CO_2 kg/tonne waste
Household	MBT plant	MSW	RCV	100%	4.3
MBT plant	Thermal treatment	Reject	L1	9%	0.1
MBT plant	Metals reprocessor	Scrap metal	L2	100%	0.1
Total					**4.5**

Treatment emissions

The emissions from each fraction are calculated in the same way as the emissions from landfill, composting and incineration. There is a slight difference for the composting phase – typically the composting period is shorter in MBT plants and so degradation will be less complete and less CO_2 will be emitted. However, emissions from the composting phase are all short term carbon dioxide with no greenhouse gas impact.

Energy use

There is a wide variation in the reported figures for energy usage, but under-reporting is common (i.e. not all energy-using processes are always included in the reported figures). A survey of Austrian plants [63] reports energy usage directly for four plants, ranging from 4 kWh per tonne of waste treated to 56 kWh/t.

Energy usage is, of course, related to the duration of the composting process, particularly the number of times the material is turned. Data made available to us indicates that an energy usage of about 0.5 kWh/tonne of input material would be required for each turning event. About 10 more additional turning events are required for long-duration treatment needed to produce a very highly stabilised waste, compared with the shorter duration processes [56]. The difference in energy use between highly stabilised and less well-stabilised residue would therefore be about 5 kWh/tonne of waste – well within the wide range of uncertainty in reported energy consumption rates outlined above. In view of this margin of uncertainty, we have not considered differences in energy requirements for alternative MBT treatments, but instead have assumed an average figure of 50 kWh of electricity per tonne of waste. At an EU-average power generation emission rate for 2000 of 0.45 kg CO_2 /kWh (see Figure 5), this equates to some 22 kg CO_2 eq / tonne of waste processed.

Displaced emissions

Displaced emissions relevant to the MBT options are heat and power replaced by incineration of RDF, and savings on emissions associated with recovery of scrap metal, using the values for ferrous metal and aluminium quoted in Appendix 7. Because of the very low CH_4 forming potential of MBT residue, we assume that no energy is recovered from landfill gas in this instance.

Displaced emissions from RDF incineration are based electricity-only recovery at a thermal efficiency of 18%, as outlined in Appendix 3. The base case assumption is that replaced power comes from average EU-power fuel mix, but the sensitivity analysis compares this situation with wind- or coal-generated electricity being replaced.

Disposal emissions

Three possible cases are considered which illustrate the wide range of possible impacts of alternative deployment approaches for MBT will have on the greenhouse gas flux from landfill disposal of MBT compost.

- **Case 1. Highly stabilised MBT compost.** About 5%[64] to 10% [61] of degradable organic carbon has been estimated to remain in highly stabilised MBT compost. For this study we adopt the results of the laboratory trials which suggest that MBT eliminates about 90% of the CH_4 forming potential of MSW. The rate of formation of the residual CH_4 is such that we assume that oxidation by micro-organisms in the landfill soil will be able to completely convert the CH_4 to CO_2. No CH_4 emission thus takes place and so there are no greenhouse gas emissions associated with landfilling of MBT residues. Remaining short-cycle carbon is assumed to be sequestered.

- **Case 2. Less stabilised MBT compost.** A shorter duration MBT process is used that results in some remaining CH_4 emission. We have simulated this by using the same CH_4 forming potential as in Case 1, but assuming that only 25% is oxidised to CO_2 by a combination of microbial oxidation and gas collection and oxidation in bio-filters, the remaining 75% escaping to the atmosphere. Flaring would have the same overall effect but MBT compost is considered unlikely to produce landfill gas with a high enough CH_4 content (ie less than about 17% by volume) to allow combustion without a pilot fuel.

- **Case 3. MBT compost used as a surface dressing for landfill site remediation or as a restoration layer, acting as biofilter, to reduce CH_4 emissions.** In these applications, decomposition of the compost continues aerobically and resistant organic matter that would have been sequestered under anaerobic conditions decomposes. In the absence of better data, we have assumed that decomposition occurs at the same rate as high-quality compost applied in an agricultural setting, as described in Appendix 4. The turnover time used (42 years) implies that 8% of the carbon in the non-dissimilated degradable carbon applied in the compost will remain in the soil outside the 100 year time horizon for sequestration. Further information on the rational for this approach, and the considerable uncertainty associated with this estimate, is given in the section on carbon sequestration in Appendix 4.

The impact of the three alternative cases for MBT compost production and use on greenhouse gas fluxes is shown in Table A4.43. Which particular case prevails will depend strongly on local conditions. It is expected that case 1 (highly stabilised compost) may be reflect the German approach to MBT, whilst the shorter treatment times used in Austria and Italy may be more typical of case 2. Case 3 can apply to either highly stabilised or less well stabilised compost, and may be the dominant approach when MBT is implemented at existing landfills, where the compost produced on site would be useful for restoring completed areas of the site. This use, however, would be unlikely to continue to provide an outlet consuming all of the compost for very long, and sooner or later we believe that the site would revert to case 1 or case 2 conditions.

The DOC of the input waste (column a of Table A4.43) is the same as described in the appendix on landfilling. The figures for the amount of DOC released during the composting stage (column b) are 90% of the figures used for traditional composting (Appendix 5). The amount of dissimilable DOC entering the landfill (column d) is therefore 10% of the DDOC for landfilling of untreated waste (see Appendix 2). In cases 1 and 3 we are assuming that any methane generated is oxidised before release, there are no methane emissions. In case 2, 75% of the methane formed is assumed to be released. All carbon degrading is released as short term CO_2 with no greenhouse impact. Sequestered carbon is calculated as shown in column f.

Table A4.43 : Estimates of carbon flows during MBT composting and landfilling of degradable fraction per tonne of waste component going to MBT composting stage.

Waste component	DOC	% dissimilated during composting	DOC to landfill	DDOC in landfill	methane generation potential kgCH_4/t waste	methane generated kg CH_4/t waste composted	methane released kg CH_4/t waste composted	short term CO_2 released kg CO2/t waste composted	Avoided energy (LFG) kg CO_2/t waste composted	Carbon sequestered kgCO_2/t waste composted
	a	b	c=a*(1-b)	d	e=d*50% *1000*16/12					f=(c-d)* 1000*44/12
Case 1. Highly stabilised MBT compost										
Paper	33%	32%	23%	1.2%	7.69	0	0	423	0.5	786
Putrescibles	19%	57%	8%	1.2%	8.02	0	0	441	0.5	251
Fines	9%	54%	4%	0.6%	3.73	0	0	205	0.5	137
Miscellaneous combustibles	28%	32%	19%	1.0%	6.52	0	0	358	0.5	665
Case 2. Less stabilised MBT compost										
Paper	33%	32%	23%	1.2%	7.69	6	6	407	0.5	786
Putrescibles	19%	57%	8%	1.2%	8.02	6	6	425	0.5	251
Fines	9%	54%	4%	0.6%	3.73	3	3	197	0.5	137
Miscellaneous combustibles	28%	32%	19%	1.0%	6.52	5	5	345	0.5	665
Case 3. MBT compost used as a surface dressing for landfill site remediation or as a restoration layer										
Paper	33%	32%	23%	1.2%	7.69	0	0	1,146	0.5	63
Putrescibles	19%	57%	8%	1.2%	8.02	0	0	672	0.5	20
Fines	9%	54%	4%	0.6%	3.73	0	0	331	0.5	11
Miscellaneous combustibles	28%	32%	19%	1.0%	6.52	0	0	971	0.5	53

Note that the data are expressed per tonne of material in question. For MSW, the emission factors are estimated from the sum of the constituent waste components multiplied by their relative proportion in the waste stream, as given in Table 9 on page 25.

OVERALL GREENHOUSE GAS FLUXES FROM MBT

Overall greenhouse gas fluxes from the various waste components and from MSW are summarised in Table A4.44, which recapitulates data presented in the separate sections of this Appendix. Fluxes for the separate waste components and whole MSW processed through MBT are shown for Case 1 (for fully stabilised compost), but for the remaining cases, only summary data for MSW is given. Note that the data refers to all steps associated with MBT treatment. Thus, for example, the CH_4 emissions attributed to paper reflect emissions from the residues contributed by paper, plus the emissions due to the 20% of incoming paper that is diverted to landfill (see Table A4.41). Similarly, fluxes attributed to metals come from the recycling stage.

The higher rate of CH_4 release assumed under Case 2 exerts a relatively modest effect on the overall total greenhouse gas flux of MBT (as shown in the right-most column of the table). For MBT with landfill of rejects, the total greenhouse gas flux increases from –323 to – 249 kg CO_2 eq/tonne of MSW, and from –216 to –141 kg CO_2 eq/tonne for MBT with incineration of rejects. The mean values from Cases 1 and 2 are used in subsequent analysis, because of the relatively slight impact, in comparison with other uncertainties in the data, that this has on the overall flux.

In contrast, the major impact of Case 3 on the overall greenhouse gas flux is very large, mostly due to the reduction in sequestered carbon that results from the aerobic decomposition of MBT compost used as a surface dressing. Case 3 is shown to illustrate the major impact of C-sequestration on the assessment, but is not pursued further in the analysis because of the anticipated limited application for Case 3 type use of MBT compost if the option becomes widely-deployed.

Table A4.44: Emission factors for wastes processed through MBT (kg CO2 eq/t material or MSW treated).

Waste management option	**Waste component**	**Short cycle CO_2**	**Fossil CO_2**				**Short cycle C sequestered**	**Sum of fossil C and sequestered C**	**CH_4 emission**	**N_2O emission**	**Total GHG flux**
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	**(GWP=-1)**		GWP=21	GWP=310	
MBT treatment with landfill of rejects and recycling of metals											
	Paper	396	0	22	-6	4	-786	-765	206.6	0	**-559**
	Putrescible	441	0	22	0	4	-251	-224	0.0	0	**-224**
	Plastic	0	0	22	0	4	0	27	0.0	0	**27**
	Glass	0	0	22	0	4	0	27	0.0	0	**27**
	Metal	0	0	22	-3038	10	0	-3006	0.0	0	**-3006**
	Textiles	147	0	22	-16	4	-503	-492	526.1	0	**34**
	Other	226	0	22	-6	4	-369	-349	206	0	**-143**
Case 1	MSW	286	0	22	-162	5	-364	-500	97	0	**-403**
Case2	MSW	276	0	22	-162	5	-364	-500	171	0	**-329**
Case 3	MSW	551	0	22	-162	5	-99	-234	97	0	**-137**
Mean of Cases 1 & 2	**MSW**	**281**	**0**	**22**	**-162**	**5**	**-364**	**-500**	**134**	**0**	**-366**
MBT with incineration of rejects in electricity only incinerator and recycling of metals											
	Paper	580	0	22	-51	4	-629	-653	0	3	**-650**
	Putrescible	441	0	22	0	4	-251	-224	0	0	**-224**
	Plastic	0	2237	22	-703	4	0	1560	0	15	**1575**
	Glass	0	0	22	0	4	0	27	0	0	**27**
	Metal	0	0	22	-3038	10	0	-3006	0	0	**-3006**
	Textiles	718	718	22	-326	4	0	420	0	15	**434**
	Other	285	63	22	-26	4	-213	-149	0	5	**-144**
Case 1	MSW	358	205	22	-241	5	-289	-298	0	3	**-295**
Case 2	MSW	349	205	22	-241	5	-289	-298	74	3	**-221**
Case 3	MSW	604	205	22	-241	5	-23	-33	0	3	**-30**
Mean of Cases 1 & 2	**MSW**	**353**	**205**	**22**	**-241**	**5**	**-289**	**-298**	**37**	**3**	**-256**

Note that the data are expressed per tonne of material in question. For MSW, the emission factors are estimated from the sum of the constituent waste components multiplied by their relative proportion in the waste stream, as given in Table 9 on page 25.

MBT Costs

No data has been found on MBT gate fees. However, information from Germany suggests that MBT costs are around 87 Euro/t, including landfill disposal fees. However, since waste management charges in Germany are usually at the upper end of the range for other member states, we have adopted lower figures for the EU as a whole. We have selected figures of 60 Euro/t for MBT with landfill and 75 Euro/t for MBT with incineration. These figures are slightly higher than the composting, AD and incineration figures to reflect the extra processing (separation) stages necessary for MBT.

Appendix 5. Composting

COMPOSTING PROCESS DESCRIPTION

Composting is the aerobic degradation of waste to produce compost which can be used as a soil improver. Although home compost heaps have been in use for centuries, large centralised plants have been established more recently. Box 7 gives a brief outline of the biological processes involved with composting.

Box 7: Carbon degradation during composting

Biochemical processes of carbon mineralisation take place through a series of enzymatically-catalysed steps. Complex bio-polymers such as carbohydrates and proteins are first broken into smaller fragments (sugars, amino acids etc) by hydrolytic enzymes secreted by micro-organisms growing on the biodegradable components of the waste. The process is accelerated by animals that live in the waste (such as fly larvae and other small arthropods, worms etc) that shred the waste into small particles that increases the surface area for enzyme attack. Animals also secrete digestive enzymes and also process the organic materials through their guts where further digestion takes place. The breakdown products are used as a source of energy and nutrients for the micro-flora and fauna growing on the waste, the overall process being to use oxygen from the air to convert the organic carbon to CO_2. The heat released from the process causes the decaying waste to heat up, and this further increases the rate of decomposition. A proportion of the carbon is cycled through the microbial biomass, and some, such as lignin (one of the main structural polymers of woody tissue) is inherently resistant to decomposition. The decomposing waste (in a compost heap, for example) thus becomes increasingly dominated by materials resistant to further decomposition.

During the decomposition process, the temperature and rates of CO_2 evolution and O_2 consumption increase to a maximum and then tail-off over the next days to weeks as the readily-degradable materials are used up. The end product of decomposition, compost, consists of materials resistant to decomposition, such as lignin and polymers formed from the remains of the biomass, along with humus. Humus is the term for highly condensed aromatic structures of high molecular mass which are highly resistant to further decomposition. Humus is the source of the humic acids that contribute to the dark colour of most soils. Soil humus plays an important role in soil fertility, as discussed later. Typically about 60 % of the mass of initial material is lost (as CO_2 and H_2O) during the composting process.

Successful composting depends on source separation of organic material to avoid contamination of the final product. Most composting schemes use mainly garden waste, although some schemes also use separately collected vegetable and fruit waste from kitchens and some can use small quantities of paper (up to 15%).

We have assumed that composting uses only source-separated waste, as most composting schemes aim to produce a usable compost. Material is either collected at the kerbside or from household waste sites. Levels of contaminants in non-source segregated composts, such as glass, hypodermic needles and plastic and metal fragments, prevent the marketing of the compost.

There are a variety of centralised composting systems. Simple ‘windrow’ systems are open-air piles which are periodically turned by agricultural machinery. In ‘semi-closed’ systems the windrows may be positioned under a covered area to reduce the effects of rainfall on the composting process. Closed systems are more complex indoor tunnel, hall or container systems where the temperature and humidity are controlled, air may be blown into the compost and the compost may be agitated automatically. The process takes several weeks or months to complete, whereupon the composted waste may be screened again to remove reject material such as large pieces of wood. The reject material from sorting is usually landfilled. Bio-filters may be used to reduce odours and bio-aerosols, and in some cases additional filters are used to control volatile organic compounds.

In centralised composting plants about 30 to 50% of the mass of incoming waste is converted to compost. The average yield of 16 closed process composting plants in Germany, the Netherlands and Denmark surveyed in a recent RVF report was 47% [65], but these plants are probably more efficient than the average European plant. For this study we have assumed a yield of 40%. In a good quality scheme where careful attention is paid to ensuring effective segregation of compostable material from other waste, 10-15% of the incoming feedstock may be rejected to landfill [66], although 20-25% can be rejected in poorly designed schemes. The plants surveyed in the report above achieved an average reject rate of only 6%. For this study we have assumed a reject rate of 10%. The remainder of the original waste is lost as moisture and carbon dioxide during the composting process.

COMPOSTING GREENHOUSE GAS EMISSIONS

The following steps are considered in the analysis:

- **Mobilisation.** This includes collection and transport of waste from the householders to the composting plant and transport of compost to market or landfill.
- **Treatment.** Carbon is released as carbon dioxide during the composting process. This is all short term carbon dioxide as it originates from degradable organic matter. Emissions arise from fuel use for turning and processing the compost.
- **Use / Disposal.** Carbon may be sequestered in soil organic carbon as a result of the use of compost.
- **Displaced emissions.** Avoided emissions from displacement of peat or fertilisers by compost.

MOBILISATION

The transport routes considered are:

household → composter → market for marketable compost
→ landfill for unmarketable compost and rejects

or household → household waste[a] site→ composter → market for marketable compost
→ landfill for unmarketable compost and rejects

For home composting there are no transport emissions.

---

[a] Household waste sites are facilities for householders to bring bulky wastes and recyclables to that are not collected by the collection service.

Vehicle emission characteristics and assumed journey distances are given in Appendix 1. Emissions are summarised in Table A5.45.

Table A5.45 Mobilisation emissions associated with composting

From	To	Load	Vehicle	% by route	CO_2 emissions kg/t
Household	Composter	Green waste	L1	67%	2.4
Household	CA site	Green waste	CAR	33%	4.2
CA site	Composter	Green waste	L2	33%	0.6
Household	Compost heap	Green waste		0%	0.0
Composter	Landfill	Rejects	L1	100%	0.4
Composter	Market	Compost	L2	100%	0.2
Composter	Landfill	Compost	L1	0%	0.0
Total					**7.7**

TREATMENT EMISSIONS

The emissions of carbon dioxide during the composting process were estimated for each of the two main waste streams which can be composted: paper and putrescible (food and garden) waste. We have assumed that currently the waste entering composting plants is 95% food and garden waste and 5% paper waste. The proportions of food to garden waste vary widely amongst composting plants, although most plants take mainly garden waste at present. We have little data on the relative proportions of food and garden waste arisings in each EU country, as this is generally reported only in aggregate form as 'putrescible waste'. However in the UK the split is approximately 55% food and 45% garden waste of which most of the garden waste is delivered to household waste sites. We have therefore modelled the input as average UK putrescible waste because this enables scenario modelling based on reported arisings for each member state. The ratio of food to garden waste makes no difference to greenhouse gas emissions as all the carbon dioxide emitted is short term.

Paper actually improves the final composition of the compost, making it more fibrous, and therefore we have assumed that by 2020 the proportion of paper will have risen to 10% as un-recyclable paper is diverted to composting or AD schemes.

Carbon dioxide emissions were based on an analysis of the carbon content of the waste, and estimates of the proportion of this carbon which will degrade. The basic methodology is the same as described for the landfill degradation process (see Appendix 2). It involved estimating '**degradable organic carbon**' or DOC and then '**dissimilable organic carbon**' (DDOC), i.e. the fraction of the DOC which actually degrades during the composting process. The dissimilable organic carbon was converted into a figure for kilograms of carbon dioxide per tonne of waste stream treated.

The estimates of degradable organic carbon content of the waste are as stated in Appendix 2 on landfilling. The estimation of the dissimilable fraction of DOC is more difficult. The fraction of DOC which dissimilates during composting could differ from that during landfilling for several reasons:

- Composting is primarily an aerobic process whereas waste in landfills degrades mainly in anaerobic conditions. This could affect the degradation of lignin in wood in particular, as lignin does not degrade anaerobically.

- In composting, material is shredded and turned to allow full access to air, water, nutrients and microbes essential for the degradation process. In landfilling, waste is not mixed after deposition and the resulting uneven distribution of moisture, nutrients and waste type often results in very uneven rates of decomposition across through the waste mass.
- The composting process takes only a few weeks or months, whereas waste resides permanently in landfill sites.

Whilst the first two points could imply higher dissimilable fractions for composting than for landfilling, the last issue could imply the opposite. In the absence of better data, we did not feel justified in selecting different DDOC values for composting than for landfilling. In any case, the values make no difference to greenhouse gas fluxes as the carbon released is all short term. The estimates used are therefore the same as for landfilling. They are shown in Table A5.46.

These estimates are supported by a mass balance based on the percentage of carbon remaining in compost. Assuming around 20% carbon per unit of dry matter and around 25% water content gives a carbon content of 15% per unit of compost. Assuming a yield of 40% compost per unit of putrescible waste with an initial carbon content of 19% implies that 32% of the original carbon content of the waste remains in the compost; the remaining 68% being lost to the air. This compares well with the figure of 64% calculated from our estimates.

Compost will continue to degrade and release carbon dioxide after application to the soil or use as landfill cover and, given long enough, essentially all the carbon will be released in this way. For present purposes, we need to consider the proportion of carbon that will be mineralised to CO_2 over the 100-year time horizon. This is calculated as part of the estimation of the amount of carbon sequestered in soil as a result of compost application, the amount sequestered being the difference between the quantity of carbon applied and the amount still remaining in the soil after 100 years. The approach to these calculations is described in further detail in the section on ‘Compost as a carbon sink’ later in this appendix. About 92% of carbon applied in compost is estimated to be mineralised to CO_2 during the 100 year horizon, with the remaining 8% being considered to have been sequestered. In the case of putrescible waste, this equates to some 22 kg CO_2 eq/tonne of waste – see Table A5.46, as difference between columns (b) and (h).

We have also assumed that the 10% rejects from compost making that cannot be sold is diverted to landfill, for use as surface dressing for site restoration, where it degrades aerobically to carbon dioxide. Methane will not therefore be produced.

Table A5.46 : Estimates of carbon dioxide released during composting and over subsequent 100-years after use of compost as soil conditioner.

Waste component	**degradable organic carbon (DOC) (a)**	**CO_2 equivalent of DOC (b)=(a)*44/12*1000**	**% of DOC lost during composting (c)**	**dissimilable organic carbon (d)=(c)*(a)**	**emission factor kg CO_2/t (e)=(d)*1000*44/12**	**% DOC mineralising during 100 y after compost use (f)={(a)-(d)}*92%**	**CO_2 produced during compost use (g)=(f)*44/12*1000**	**emission factor kg CO_2/t (h)=(g)+(e)**
Paper	33%	1209	35%	12%	423	20%	721	1144
Food (55%)	15%	539	75%	11%	404	3%	124	528
Garden (45%)	24%	880	50%	12%	440	11%	404	844
Putrescibles	19%	692	64%	11.5%	420	7%	250	670

Energy use

Energy use during composting varies depending on the type of process. For a simple open windrow process the average energy consumption was estimated at 5 litres of diesel per tonne of waste, used to fuel the tractors that shred and turn the windrows. This corresponds with emissions of about 13 kg CO_2 eq/tonne. For a closed process the energy consumption was estimated at 40 kWh electricity per tonne of waste, or 18 kg CO_2 eq/tonne at EU-average power emission factor. This is the average of the 16 plants surveyed in the RVF report [65]. The higher energy use of closed plants reflects the use of gas cleaning systems to remove odour emissions as well as the electricity used for aerating the piles and maintaining correct temperature and humidity.

Differences between treatment options

In terms of greenhouse gas impact the different centralised treatment options (open, closed, semi-closed) differ only in energy use, although there are cost differences and differences in local environmental impacts (odours and bio-aerosols).

There is very little data available on home composting in the EU. Home composting has the advantage of saving transport costs and emissions, particularly for green waste which is very bulky. However, due to sub-optimal conditions in some home compost heaps, a little methane may be produced although some sources suggest that due to the small size of domestic compost heaps most of this would be oxidised before reaching the surface of the heap [67]. In the absence of any data, we have assumed that no methane is produced but we have performed a sensitivity test in which 1% of the degradable carbon is released as methane and the remainder as carbon dioxide. This increases the greenhouse gas flux of the process from zero to 51 kg CO_2 equivalent per tonne of putrescible waste treated.

USE / DISPOSAL AND DISPLACED EMISSIONS

After the compost is produced and applied land, it continues to degrade, releasing more carbon dioxide and forming humic compounds that then mineralise much more slowly than the organic matter originally applied in the compost. Compost applications may therefore increase the store of soil organic matter. However, it is only carbon applied in any given year that is stored in the soil for longer than the time horizon adopted for global warming (100 years) that can be considered as having been sequestered. This sink should be accounted for in the calculation of greenhouse gas emissions as a negative contribution to the emissions total. Our approach to this important issue is discussed separately in the following section.

The use of compost may also displace peat or inorganic fertilisers, with associated emissions benefits. This topic is discussed in the final section of this appendix.

COMPOST AS A CARBON SINK

The IPCC has identified carbon sequestration in soils as one of three carbon mitigation measures for agriculture, the other two options being a reduction in agriculturally related emissions and the replacement of fossil fuels with biofuels [68].

The loss of organic carbon in soils has been one of the major environmental consequences of industrial agriculture. Arable top soils in Europe commonly contain 1-3 % of organic carbon.

ECAF [69] suggest that over about 20 years tillage, most agricultural soils will have lost about 50% of their organic carbon. This is largely because of the reliance of industrial agriculture on inorganic fertilisers, rather than organic composts and manures, as a source of crop nutrients, and the extensive use of tillage. Thus the input of plant-derived materials that break down to form soil organic matter is reduced, whilst high rates of tillage increase the exposure of the organic matter to erosion and loss through oxidation. The annual net release of carbon from global agricultural activities is about 800 Mt/year, or about 14% of current fossil fuel burning [70].

Compost may provide a useful source of soil organic matter, contributing to improvements in soil physical structure and fertility (see Box 8). Some regions are particularly short of soil organic matter, for example, soils in the Mediterranean basin. A recent study review concluded that organic waste re-use through compost applications seems to be the best way of improving the organic matter status of such soils [71]. Certainly there is a great deal of work that has demonstrated the value of compost in improving soil fertility and crop yields.

Estimating carbon sequestration in soils

The IPCC framework for estimating greenhouse gas fluxes is based on a 100-year time horizon. Organic carbon stored in soil as a result of compost application will therefore only be considered to have been sequestered if it remains locked up in the soil for at least 100 years. Estimating a precise lifetime for soil organic matter derived from compost addition is very difficult, because of the large number of inter-converting pools of carbon involved, each with its own turnover rate, which is in turn determined by local factors such as soil type, temperature and moisture (see Box 9). The investigation of soil organic matter transformations is a highly active research area, because of the pivotal role of soil organic matter in determining fertility, agricultural productivity and environmental impacts. Much of the work has been undertaken though long-term field experiments and simulation modelling.

Box 8: The role of soil organic matter.

Soil organic matter consists of a whole series of products which range from undecayed plant and animal tissues through ephemeral products of decomposition to fairly stable brown to black material that is normally defined as humus [72]. Soil organic matter, defined as the totality of the organic matter in soil, also includes the organisms that live in the soil, the soil biomass, although they usually account for less than 5% of soil organic carbon.

The functions of soil organic matter are complex, but may be summarised as follows [73]:

- **Energy for soil organisms.** Inputs of organic matter provide energy and nutrients for soil organisms, which are essential for soil structure and fertility, and for increasing the population of beneficial soil animals, such as earthworms, which then mix and aerate the soil.

- **Soil structure and stability.** Certain types of organic substances in the soil, such as microbially produced polysaccharides, cement soil particles together to form crumbs, resulting in the development of good soil structure (or ‘tilth’). The crumb structure helps to keep open aeration channels and improves water holding properties, promoting efficient gas exchange and helping root penetration. Light sandy soils therefore become more aggregated and heavy clay soils become more porous.

- **Water relations.** Improvements in soil structure resulting from organic matter additions improve the permeability and drainage, with less likelihood of capping and slaking. Soil organic matter can absorb and hold considerable quantities of water, much of which is then available to crop roots.

- **Erosion control.** A further benefit of good soil structure is greater resistance to erosion, either by wind or run-off water.

- **Availability of plant nutrients.** Microbial mineralisation of manures and composts applied to the soil releases variable amounts of nutrients, particularly N, P and S. In addition, humus contains numerous negatively-charged sites which, along with those on the clay minerals, provide about half of the cation exchange capacity (CEC) of the soil. This provides an important mechanism by which the leaching out of positively-charged ions may be reduced. Some of these ions may be important plant nutrients – such as ammonium (NH_4^+), potassium (K^+), magnesium (Mg^{2+}) and calcium (Ca^{2+}) ions. The mechanism also provides a means of buffering the acidity in soil and reducing changes in soil acidity.

- **Other effects.** Other effects of soil organic matter include influences on crop growth, incidence of pathogens and diseases and absorption of pesticides [72].

Box 9: Soil organic matter dynamics

Soil organic matter is derived from the remains of plants growing in the soil or imported from elsewhere as manures or composts. The decomposition of the plant material results in the formation of soil organic matter, and the release of CO_2. Decomposition of plant remains in soil usually occurs in two distinct phases. During the rapid first phase, the substrate is broken down and secondary products, the soil microbial biomass and its metabolites, are synthesised concomitantly. This microbial biomass and its metabolic products are themselves the substrates for the much slower second phase. The remaining organic materials are converted to high molecular weight condensed polyaromatic compounds known as humin, humic and fulvic acids (collectively known as humic substances). Humic substances are mostly closely associated with clay minerals which confers resistance to decomposition. Humic substances have ages ranging from 500 – 2000 years, indicating their high degree of resistance to biological decay. For a review of soil organic matter dynamics, see reference [74].

Plant materials differ widely in their rates of breakdown in the soil. Materials with a wide C/N ratio and those that are rich in polyphenols decompose relatively slowly. Lignin, the principal natural polymer in wood, is particularly resistant to decomposition. There are relatively few micro-organisms that can degrade lignin and they are exclusively aerobic: indeed the lignases that they produce are thought to require hydrogen peroxide for their operation [75]. Not only is lignin itself resistant to decay, but it encrusts the cellulose and hemicelluloses of plant cell walls, impeding the decomposition of these materials also. Nevertheless, lignin is degraded steadily in aerobic soils – some 30% of radio-labelled lignin was converted to CO_2 when incubated with moist aerobic soil for a year [76].

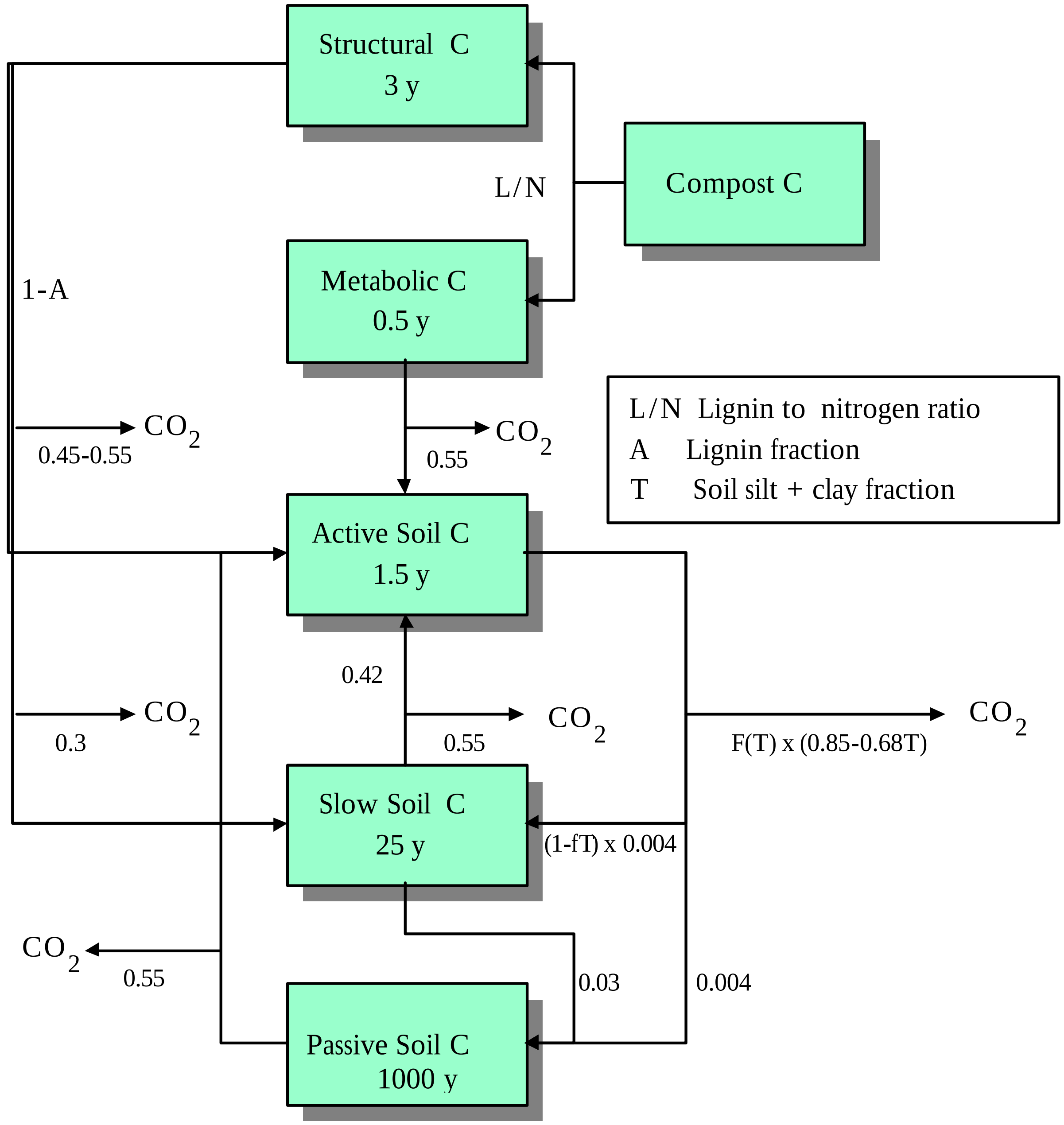
The study of soil organic matter transformations has been considerably advanced from the results of long-term (decades) field experiments, coupled with computer simulation modelling. Good examples of such studies are the Rothamsted experiments in the UK (which have been running since the 1850s) [74] and the more recent Ultuna field studies in Sweden [77]. Models based on these studies have been described by Jenkinson [74] and Paustian et al [77]. Further studies on soil organic matter dynamics were the subject of a recent symposium [78]. An example of soil organic matter modelling is provided by the CENTURY model [79], applied to data from the Ultuna experiment The various pools of soil organic matter used by this model are shown in the diagram (redrawn from [77]), along with the estimated turnover times. The steady state concentrations of active, slow and passive soil organic matter at the Ultuna site were estimated as 3, 47 and 53 % respectively. The numerical values indicate the relative partitioning of C between the various organic matter components and CO_2. Only a very small proportion of input carbon ends up in the long-lived 'passive' pool. Note that there is a great deal of debate as to exactly what substances make up the operationally-defined pools of C considered in the model. The model has proven to be reasonably successful in predicting long-term changes in soil C following organic matter additions.

Key factors affecting soil organic matter are:

- type of organic material – in particular, the amount of resistant organic matter such as lignin, and the carbon to nitrogen ratio;
- the method of application of the organic matter – e.g. surface mulch or incorporation below the soil surface;
- temperature and moisture regime (decay is slower in cool, wet climates than warm, moist climates);
- tillage practices – decay increases with tillage;
- availability of clay minerals to form complexes with humus (clays stabilise organic matter).

These factors are highly specific to particular situations and therefore the extent of carbon sequestration will vary markedly with conditions and from site to site.

Figure A5.37: Soil organic matter dynamics (redrawn from reference 77).



For the purposes of this study, we are interested in how much carbon in compost produced and applied in *a given reference year* is retained in the soil for longer than the 100-year time horizon for carbon sequestration. Although a great deal of valuable information now exists on the turnover of soil organic carbon, the question of assigning a typical average value for the persistence of carbon applied to soils in compost remains highly problematical.

Life times of various pools of soil organic carbon ranging from 20 to 2000 years have been proposed for bounding purposes by the US EPA [67]. Isotope studies [80] have shown that turnover times differ widely for different fractions of soil organic matter, ranging from less than a year for microbial biomass to between 5 and 1000 years for organic matter associated with silt and clay particles. Obviously life times towards the upper limit of these ranges for compost are implausible, since this would imply virtually no decomposition within the 100-year time horizon under consideration. Modelling studies indicate that very little added organic carbon is incorporated into the very long-lived carbon pool in soils – see Figure A5.37. Therefore for the purposes of this study we need information on the life time of carbon shortly after it enters the soil organic matter pool.

Other work has indicated that the average lifetime of soil organic carbon is well towards the lower end of these ranges. For example, Jenkinson [72] calculates an average global turnover time for soil organic matter of 27 years[a]. Turnover is much faster in tropical than in temperate soils. Applying first-order decay kinetics (which are widely used in this field), this global average suggests that only 2% of the carbon applied to soil today would remain in the soil organic matter in 100 years time. Long-term field experiments are, to a first approximation, broadly consistent with this global average. For example, the turnover time of soil organic carbon added in farmyard manure (at 3 tC/ha/y for 125 years) was estimated at 42 years, with comparable data for soil nitrogen (the turnover of which is closely linked to that of carbon) being 18-36 years[72]. The amount of carbon or nitrogen remaining after 100 years, assuming first-order decay, would therefore be 9% for carbon, or 0.4-6.2% for the nitrogen data.

Smith and colleagues [81] have reported on the scope for carbon sequestration in soils resulting from changes in land management practices to mitigate European greenhouse gas impacts. They report summary results from several long-term field experiments in which sewage sludge, animal manure and cereal straw had been incorporated annually into the top layer of soil. They derived linear correlations between the organic matter addition and the annual increase in soil organic carbon, which, if a steady state has been reached between carbon input and soil organic carbon, allows the amount of added carbon surviving in the soil for a given period to be calculated. Using the equations provided by Smith et al and assuming a carbon content for manure and sewage sludge of 20% results in estimates of 9 and 6 % as the proportion of added carbon persisting for over 100 years. Similar calculations with straw (at an assumed C content of 35%) indicate that 22% persists in the soil for 100 years. Straw may be expected to persist longer than the other organic materials because of its higher lignin content. It is important to note, however, that incorporation of carbon into the soil organic matter is dependent on the substrate being mechanically introduced below the soil surface, rather than just being applied as a surface dressing or mulching layer. Surface applied material will tend not to be incorporated into the soil organic matter and will mineralise much faster [81].

Other studies have shown much lower levels of persistence of carbon added in soil conditioners and manures. For example, Cortellini et al [95] report the effects of adding compost (made from straw and sewage sludge) or sewage sludge to soils at up to 15 t dry matter per year for six years. They found that by the end of the experiment, the soil organic matter had increased by up to 13% compared with unmanured soils. From their data, it appears that about one fifth of the carbon applied annually as sludge or compost persisted in the top 40 cm of soil for the six years of the experiment. This would equate to a turnover time of about 2 years – ie virtually none of the added carbon would be expected to persist for 100 years. Other authors have detected improvements in soil organic carbon content, but only with massive rates of application [82].

---

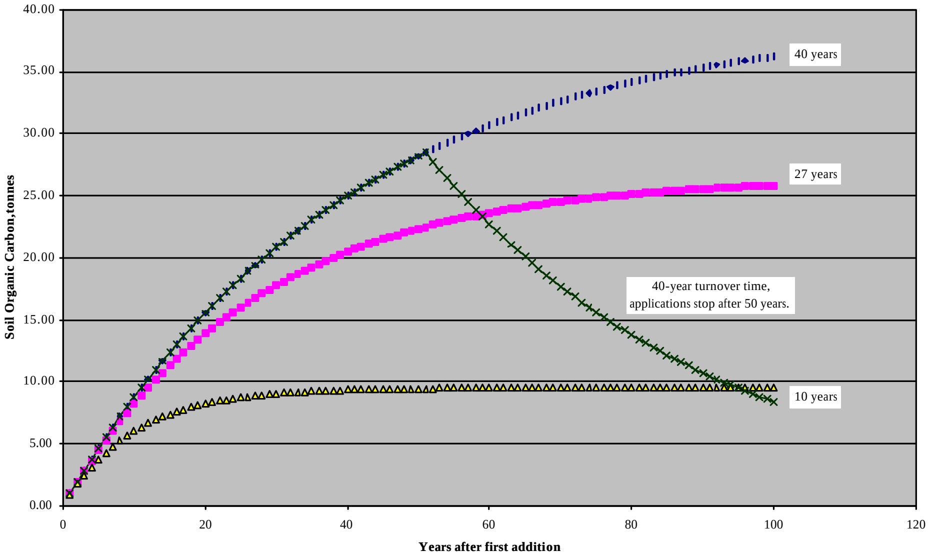
[a] Based on estimates of net CO_2 fixation by terrestrial plants of 55 x 10^9 tonnes / year and an estimated pool of 1500 x 10^9 tonnes of organic carbon stored in soil. The persistence of soil organic carbon can also be expressed as a half life. This is the time needed for half of the added organic carbon to mineralise to CO_2 . Half life is related to turnover time as follows: half life = turnover time * $\log_e 2$.

For the purposes of this study, we have adopted a compost carbon turnover time in soil of 40 years. The turnover time selected approximates to the 42 years estimated from the Rothamsted carbon data. It corresponds with 8.2 % of added carbon persisting in the soil 100 years after application, the remaining 91.8 % being mineralised to CO_2. The ~8 % sequestration rate is broadly consistent with the 100-year storage rates of 6-9% calculated above from Smith's data for manure and sewage sludge, but less than the 22% for straw incorporation. In the latter case, the high lignin content of the straw is expected to contribute to its relatively slow breakdown in soil. The adopted rate is much higher than the estimates based on the Cortellini study, where virtually no carbon would be stored 100 years after applications of compost or sewage sludge. We believe the adopted sequestration rate to be a reasonable estimate for compost incorporated into the soil through tillage. However, if compost is surface applied only, then lower rates of sequestration will result. The study makes no correction for surface as opposed to sub-surface incorporation of compost, and so it is expected that the adopted rate will in fact tend to overestimate carbon storage through this mechanism.

A further factor which will influence carbon sequestration, but which it has not been possible to include in this analysis is the impact of climate change. Most climate change models suggest that Europe will become warmer and drier during the next 100 years, particularly in Mediterranean regions. Increases in soil temperature and aridity will accelerate soil organic matter mineralisation rates. This will have the effect of increasing the need for remediative measures, such as compost application, if fertility is to be maintained, but at the same time the faster turnover of organic matter will reduce the extent of carbon storage in soils.

Although not directly relevant to the issue of C storage from compost applied in a given reference year, it is interesting to consider how repeated applications increase soil organic carbon concentrations, with all the consequential benefits of improved soil stability, fertility and water holding properties discussed earlier. This is illustrated in Figure A5.38 for compost with an assumed turnover time of 40 years, together with alternative values of 27 and 10 years. In these last examples, less than 2% of the carbon added in year zero would remain in the soil for more than 100 years. Also shown is the case where annual addition of compost ceases after 50 years, for the 40-year turnover time example.

Figure A5.38: Accumulation of soil organic carbon following annual applications of 1 tonne C / ha in compost with an assumed turnover time of 40, 27 or 10 years.



The carbon added each year (a) decays according to the equation

$$y = a.\exp{-kt},$$

where y is the amount of carbon remaining after t years and k is the first-order decay rate constant (ie 1/turnover time). In the example given, compost is applied annually at a rate of 1 tonne of carbon per hectare. The total amount of carbon stored in the soil increases steadily until the annual increment in new compost carbon just equals the amount of carbon mineralised in that year from all previous additions. (For simplicity, the original carbon content of the soil has been set at zero.) For the 10 year turnover time, the soil carbon content reaches a plateau of just under 10 times the annual input in compost after about 40 years applications, but for longer turnover times of 27 and 40 years, soil organic carbon is still increasing, even after 100 applications. The figure also shows the effect of stopping compost applications after 50 years, in the case of the 40 year turnover time. The accumulated carbon immediately begins to decline, reducing to about 50% in the next 30 years – a rate of change similar to that recorded in agricultural situations in many parts of Europe [69].

Whilst highly simplified, the above model is considered to provide a reasonable generalisation for present purposes, given the absence of much of the key data required for a more sophisticated analysis.

Armed with an estimate of the average carbon sequestration rate for compost, we can now express this in terms of CO_2 equivalents per tonne of compost. Compost typically contains about 180 kg C carbon by weight /tonne on a fresh weight basis – see Table A5.48 on page 154 **From the assumed sequestration rate (8.2%), this equates to a storage term of 54 kg CO_2 per tonne of compost, or some 22 kg CO_2 per tonne of putrescible waste prior to composting.**

We will now consider the remaining displaced impacts associated with compost use – namely those of avoided peat and fertiliser.

Displacement of peat and fertilisers by compost

The use of compost may replace other materials or alternatively new markets may open up to absorb compost in its own right, rather than as a replacement for other materials. If other materials are displaced by compost, we need to consider what impact this replacement has on overall greenhouse gas fluxes. Two classes of material may be displaced by composts:

- plant growing media and soil conditioners (including peat);
- inorganic fertilisers, which may be at least partly displaced by nutrients in the compost.

Plant growth media and soil conditioners

Typical applications of compost are given in Table A5.47. The traditional material used as a growing medium and as a soil conditioner is peat. The carbon in peat is derived from plant remains laid down since the last glaciation some 10,000 years ago and so will be treated as fossil carbon. The use of peat results in the mineralisation of this carbon, leading to the release of CO_2, so making a net positive contribution to global warming. Alternatives to peat are also in use, such as forest bark for mulching, coconut fibre (coir) and other plant remains of contemporary origin. Since the substitution of compost for these other materials is unlikely to have a significant impact of greenhouse gas fluxes, we will consider only the case where compost replaces the fossil-derived carbon in peat.

Table A5.47: Applications and typical uses of compost [83].

Application	Typical uses
Hobby gardening	Potting mixes as partial or complete replacement for peat
Commercial horticulture	Growth medium for container-grown ornamentals (horticultural crops such as cucumbers, tomatoes and peppers are mostly grown in hydroponic systems nowadays)
Fruit growing/viticulture	Compost for filling planting holes
Agriculture	Mulching (compost applied to the surface of the soil to help water retention and to reduce weed emergence). As a soil conditioner to improve soil organic matter content.
Landscaping	Filling material for tree planting, media for roof gardens, public gardens and parks.
Building industry	In-filling material for site restoration after construction work is completed
Local authorities	Garden and landscaping applications.
Land rehabilitation	As replacement or cover material for contaminated soil

There are about 25 Mha of peatlands in the EU, mostly in Scandinavia, Ireland, northern Britain and Germany[84]. About 53 % remains in its natural state, with 31% devoted to forestry and 15% to agriculture. The remaining 1% is extracted for use in the peat industry. Of this extracted peat, some 85% is burnt as a fuel (in Ireland and Finland) and the rest is used in horticulture, agriculture and land reclamation, plus smaller scale uses such as a bio-filtration medium [85]. EU-wide use of peat as a growing medium and soil conditioner is estimated at about 18 million m^3 per year [86]. At an average bulk density of milled peat of 300 kg/m^3 (range 200 – 400 kg/m^3 [84]), this equates to some 5.4 Mtonnes /year.

Peat will tend to be replaced by compost on a volume rather than mass basis. Therefore if all of the peat used in these applications were displaced by compost, some 18 million m^3 would be required, or about 12 Mtonnes of compost - assuming an average bulk density of 0.68 $tonnes/m^3$ (taken as the mid point of 0.548-0.807 $tonnes/m^3$ quoted by the UK DoE [99]). It is unlikely that all peat used would be replaced by compost.

In a peat bog, the waterlogged conditions lead to establishment of an anaerobic environment in which plant material is inhibited from breaking down. Large amounts of carbon are stored in the world's peat bogs. However, when the peat is extracted it becomes exposed to aerobic conditions. Peat used as a growing medium or soil conditioner mineralises rapidly, releasing carbon as CO_2. Although a small fraction of the carbon from peat will be sequestered in stable humic compounds in the soil, we will assume that essentially all is mineralised within 100 years. This is supported by a study of carbon decay rates in peat [87] which implies a half life of 12 years for carbon – meaning that 99.5% of the carbon would have decayed within 100 years. The use of compost therefore saves the emission of fossil CO_2 equivalent to the carbon content of the displaced peat.

Peat contains around 50% carbon (on a dry weight basis), which at a moisture content of 55% [9] is 23% carbon, and has a density of around 300 kg/m^3 [84]. If all this carbon is released during mineralisation of the peat, this equates to 247 kg CO_2 $/m^3$. Each cubic metre of peat replaced by compost will therefore save the emission of about 247 kg of CO_2, equivalent to about 362 kg CO_2 per tonne of compost.

In order to estimate actual emission savings, we need to consider the use of the MSW compost to see what proportion of MSW compost produced in the EU might displace peat. This figure is likely to vary widely across the EU, given the wide variation in the use of peat. Usage of peat in horticulture (as opposed to for fuel) is highest in Germany, the Netherlands, the UK and Finland (Figure A5.39). In contrast, much less peat is used for horticulture in Italy, Spain and France, which, apart from Germany, have the highest rates of compost production.

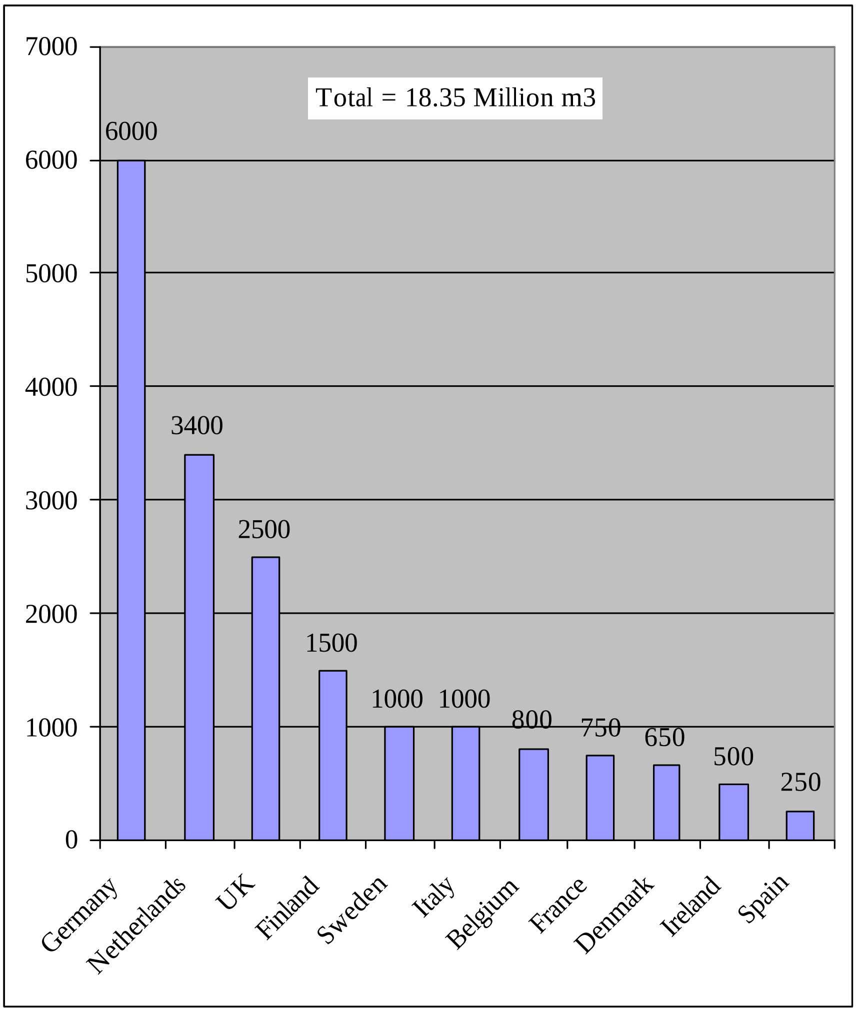
Compost is most likely to replace peat in hobby gardening and horticulture applications, where peat is used primarily for growing container plants and pot plants. In Germany around 12% of green waste compost is used in hobby gardening and 15% in commercial horticulture [83], and this amount (27%) could in theory displace peat. We have assumed that on average in the EU 20% of MSW compost goes to horticultural use and displaces the use of peat. **This results in a saving of 29 kg CO_2 per tonne of waste composted (at a yield of 0.4t compost per tonne of green waste). For home composting, the same credits have been assumed.** Although all of home-produced compost is likely to be used in the garden, we have assumed that it is used primarily as a mulch or soil improver, with only limited displacement of peat (20%).

Avoidance of peat has several other greenhouse impacts which have been **excluded** from this study. These include:

- Changes in emissions of methane and CO_2 when mature peat bogs are harvested.
- Emissions from transport and harvesting of peat.

- N_2O emissions from peat bogs.

Figure A5.39: Horticultural peat use in western Europe in 1999 [86].



The first of these impacts is the most important and is discussed in the following paragraphs. Undisturbed mature peat bogs accumulate carbon from the atmosphere through photosynthesis as peat bog plants grow. This results in a net flux of CO_2 from the atmosphere to the peat. However, mature bogs release some of the carbon back to the atmosphere as CH_4, with a global warming potential 21 times that of CO_2 (over 100 years). This is generally believed to outweigh the carbon dioxide flux, thus making mature peat bogs a net contributor to global warming.

The net greenhouse gas emissions of mature peat bogs change if the peatlands are drained and the peat harvested. Uptake of CO_2 is replaced by emission of CO_2 as stored carbon is oxidised in the aerobic conditions, but emissions of methane are reduced and emissions of N_2O are greatly reduced or stopped. However, sources differ as to whether this leads to a net increase or decrease in global warming potential.

For example, the greenhouse gas impacts of peatlands in Finland have been reviewed recently [88]. The mean CO_2 uptake for Finnish peat bogs is 75 g CO_2 /m^2 /y. The mean flux of CH_4 from Finnish peat bogs is about 13 g CH_4 / m^2 /y, equivalent to 273 g CO_2 / m^2 /y, although the variability ranges from 0 to nearly 30 g CH_4 / m^2 /y. The peat bogs therefore have a net global warming contribution of around 200 g CO_2 equivalent/m^2/y. A review for the Irish power production company Bord na Mona also claims global warming benefits from drainage of mature peat bogs [84].

Figures for an Irish peat bog from the ExternE study [9] give quite a different picture. Before harvesting, the bog has an uptake of 19 g CO_2 / m^2 /y and emissions of 1.8 g CH_4 / m^2 /y and 3.9 mg N_2O/m^2/y, giving a net flux of 20 g CO_2 equivalent/ m^2 /y. After harvesting however, although methane emissions are reduced to 0.47 g CH_4 / m^2 /y and N_2O emissions to zero, there is a large emission of 240 g CO_2/m^2/y which gives a net flux of 249 g CO_2 equivalent/m^2/y. Harvesting the peat has therefore created an additional flux of 229 g CO_2 equivalent/m^2/y. A similar study for Finland in the same reference also cites positive greenhouse gas fluxes from the drainage and extraction of peat.

Apart from this variability in estimates of the changes in greenhouse gas emissions during peat bog drainage and harvesting, an assessment should include consideration of the fate of the bog after harvesting. If the bog is allowed to revert to its natural state through re-flooding, it has been argued that carbon could be sequestered through re-accumulation of the peat [84]. However, it will take thousands of years for the bog to reach its former thickness. Global warming benefits could also be attained through carbon sequestration if the bog is afforested after use. However, if the bog is used for agriculture there will be net greenhouse gas emissions [88].

Given the uncertainty in these factors, we have excluded consideration of greenhouse gas fluxes from mature undisturbed peat bogs from the present study.

Ecological impacts

The drainage of peat bogs has important adverse ecological impacts arising from the destruction of an increasingly rare habitat [e.g. 89]. Even if the bog is re-flooded, it will take thousands of years to re-establish the same ecological conditions. Afforestation of the bog with a commercial plantation, or conversion to agricultural use, will reduce biodiversity. The use of compost as a peat replacement may therefore be beneficial in terms of ecology. However, consideration of this effect is outside the scope of the present study as we are only looking at greenhouse impacts.

Fertilisers

We will now consider the potential of compost to replace the use of mineral fertilisers. Compost contains significant concentrations of the three plant macro-nutrients nitrogen (N), phosphorus (P) and potassium (K). Although the concentrations in compost are low in comparison with inorganic fertilisers, they may never the less be of value to crops and reduce the need for inorganic fertiliser applications. In the case of N, the plant nutrient required in greatest quantities, nearly all of the N present in compost is incorporated into organic compounds. This N only becomes available for uptake by plants after micro-organisms have converted the organic N into inorganic forms, namely ammonium (NH_4^+) and nitrate (NO_3^-) ions.

Nitrate and NH_4^+ are usually supplied to crops in the form of inorganic fertilisers such as ammonium nitrate (‘Nitram’ – 34.5% inorganic N) or urea (47% inorganic N). Inorganic fertilisers provide a source of N to the crop that is therefore readily available at the time required by the growth of the crop. However, there are several potential adverse impacts of inorganic fertiliser use:

- **Groundwater pollution**: inorganic N supplied in excess of the crop's immediate needs may be leached out of the soil by infiltrating water, threatening contamination of groundwater resources.
- **N_2O release**: Denitrifying bacteria convert NO_3^- to N_2O at anaerobic micro-sites in the soil, so contributing to the greenhouse effect (as described later).
- **Emissions from fertiliser production:** production of fertilisers is very energy-intensive, causing CO_2 release from fossil fuels, and production of N fertilisers also causes release of N_2O.

National and community legislation places restrictions of the use of high N fertilisers and manures to reduce the risk of groundwater contamination and denitrification (nitrates directive). Although groundwater pollution has no greenhouse impact and will not be considered further here, the other two factors do have greenhouse impacts. Below we discuss the emissions saved from avoided production of fertilisers. Denitrification impacts are harder to quantify and are discussed in the next section (indirect greenhouse gas effects of compost use).

There is a developing literature on the extent to which compost N may substitute for inorganic fertiliser N. Important factors include the amount, timing and repetition of applications, the type of crop and soil conditions. The quality of the compost is of course important – insufficiently stabilised compost can have very high carbon to nitrogen ratios (over 30:1), which results in 'locking up' the nitrogen in the soil due to stimulation of microbial activity. For good quality compost, results are quite variable. A recent study [90] in the USA on the growth of a grass (tall fescue) in a chamber experiment showed that compost could substitute for about one third of the N-fertiliser requirement, because the N in compost is only partially available during the first year of application. Longer term field studies have reported similar findings. For example, Baldoni et al [91] found that compost N utilisation by crops was about 20% of that of urea, during a six-year study with annual applications to a three-course crop rotation (maize-wheat-sugar beet). Other studies have shown greater benefits of compost. For example, Rodrigues et al [92] showed that single applications of either composted source-separated MSW or sewage sludge compost (at 50 and 100 tonnes/ha) at the start of a two year field experiment resulted in no difference in crop (wheat) quality or yield compared with ammonium nitrate applications of 75-150 kg N/ha. These authors concluded that good quality compost was able to enhance crop production even on a fertile soil. Because the high application of compost (100 tonnes/ha) did not result in a significant marginal increase in production, applications up to 50 tonnes/ha were suggested to avoid 'luxurious' build up of N in the soil, with potential risks of groundwater contamination. In the long run, soil fertility is expected to build up with the application of composts and so application rates may then be reduced [93, 94].

It is clear from the above that compost may replace the need for at least some of the mineral N fertiliser that would otherwise have to be supplied from inorganic fertilisers on a year-to-year cropping basis. It is unlikely, however, that all of the fertiliser demand would be replaced in the short term, at least under conventional farming practices. As mentioned above, it takes several years to build up readily available supplies of N in soil through compost addition due to the need for mineralisation reactions to make the N in compost available to the crop. Farmers are used to applying inorganic fertilisers at the time of maximum growth of the crop. Also, conventional farmers tend to regard compost primarily as a soil improver and not as a replacement for inorganic fertilisers. There is little information available to farmers about the potential for reducing inorganic fertiliser application as a result of compost addition, so most farmers tend to continue with their normal fertilising regime. Better information and awareness campaigns might help to improve the exploitation of the fertiliser benefits of compost in future years.

However, the position may well be different following the establishment of the long-term use of compost in agriculture, where a higher reserve of organic nitrogen is established. Under such circumstances, it is possible that a much greater proportion of compost nutrients may displace inorganic fertiliser. We have therefore undertaken this analysis on the basis that all of the compost nutrients can displace mineral fertiliser.

The greenhouse gas emissions associated with fertiliser manufacture that are avoided through the use of composting may now be calculated. First, we need to know the typical nutrient contents of compost. This is shown in Table A5.48, the last line of which shows average values used in this study. It should be emphasised that nutrient analyses of compost are highly variable, and higher and lower average values than those shown here have been reported elsewhere.

Table A5.48: Compost nutient concentrations, kg nutrient / tonne fresh weight of compost.

Compost description & reference*	C_{org}	N	P	K
Compost from dewatered sewage sludge & wheat straw 9:1 [95]	*88*	*16.1*	*0.7*	*0.6*
MSW compost [96]	-	8.1	-	1.5
Greenwaste compost [96]	-	6.8	-	7.8
MSW compost [97]	194	4.4	1.1	1.8
Garden waste compost [98]	175	4.7	1.3	4.0
Source separated and greenwaste composts – means [99	170	7.0	3.6	7.6
Average of non-sewage sludge composts	**180**	**6.2**	**2.0**	**4.5**

Data was calculated on a fresh weight basis from the references cited.

Secondly. we need to estimate greenhouse gas emissions associated with fertiliser manufacture. These include:

- CO_2 emissions from fossil fuels used to generate energy for the process;
- CO_2 emissions from use of natural gas as a feedstock for ammonia production;
- in the case of N fertilisers only, the N_2O emissions from nitric acid manufacture.

Energy related CO_2 emissions have reduced significantly since the 1970s due to process improvements and the greater use of natural gas as energy source. Feedstock emissions have increased slightly due to process optimisation.

Nitric acid manufacture for production of ammonium nitrate fertiliser is a major industrial source of N_2O emissions, accounting for just over a quarter of industrial emissions of this gas of 313 ktonnes in the EU in 1994 [100]. Nitric acid is manufactured by the catalytic oxidation of ammonia, during which N_2O is produced as a by-product. Emission factors of 2-9 g N_2O / kg of nitric acid produced have been quoted, and the French authorities have set emission limits of 7 g N_2O / kg of nitric acid produced for new production facilities [100].

Kongshaug [101] provides data on the CO_2 and N_2O emissions arising from the manufacture of fertilisers in the EU. It is assumed that the EU average production of N_2O from nitric acid manufacture is 0.03 t N_2O /t N, corresponding to 9 t CO2 eq/t N. However, this could decrease dramatically in future years due to new abatement processes giving 70-85% N_2O reduction. The combined CO_2 and N_2O emissions are shown in Table A5.49. Production of P fertilisers releases energy due to the exothermic reactions involved in modern sulphuric acid processes. The difference between the EU average and best available technology (BAT) emissions illustrates the potential for reductions in emissions over the next decades. We take the EU average emissions for our analysis.

Table A5.49: CO_2 equivalent emissions from mineral fertiliser manufacture

Fertiliser	MJ/kg nutrient *		CO_2 equivalent kg/t nutrient		CO_2 equivalent kg/t element	
	EU average	Best	EU average	Best	EU average	Best
N	35.3	1.82	5.29	2.45	5.29	2.45
P as P_2O_5	36.2	1.87	0.22	-0.59	0.52	-1.38
K as K_2O	11.2	0.58	0.31	0.11	0.38	0.13

We can now estimate the amount of greenhouse gas avoided if nutrients in compost completely displace the use of mineral fertiliser. This is shown in the right-most column of Table A5.50. It is believed that the uncertainties in the emissions estimates for fertiliser manufacture are small in comparison with the wider question as to the extent to which compost displaces these materials – both in terms of the nutrient content of the compost and the extent of substitution. Overall, it can be seen that emissions totalling a maximum of 35.5 kg CO_2 equivalent could be avoided per tonne of compost used.

In order to estimate actual emission savings, we need to consider the use of the MSW compost to see what proportion of MSW compost produced in the EU goes to agricultural use, thus displacing the use of fertilisers. In Germany, one reference states that around 23% of MSW compost went to agricultural use in 1992 and this amount was considered unlikely to increase for various economic and regulatory reasons [83]. In southern European countries the potential for using compost to enrich severely depleted agricultural soils could be much higher. We have assumed that on average in the EU 50% goes to agriculture (plus a further 20% to horticulture and 30% to other uses such as landscaping or restoration). **This equates to emission savings of 7.1 kg CO_2 equivalent per tonne of waste composted (assuming a compost yield of 0.4 tonnes compost per tonne of waste).**

Table A5.50: Potential greenhouse gas emissions avoided in fertiliser manufacture if compost displaces mineral fertiliser.

Nutrient element	kg CO_2 equiv/kg element	Nutrient content in compost kg/tonne fresh weight	Avoided emission kg CO_2 equiv / tonne of compost (1:1 replacement)
N	5.29	6.2	-32.8
P	0.52	2	-1.0
K	0.38	4.5	-1.7
Total			**-35.5**

Indirect greenhouse gas effects of compost use

Two indirect effects of compost use may have greenhouse gas impacts but have not been included in the present study. These are:

- Impacts of a switch to organic farming techniques
- Reduction of N_2O emissions from denitrification in soils

Benefits of organic farming

High rates of compost or green manure (i.e. crops grown to fix N which are then incorporated into the soil) application are associated with ‘organic’ farming techniques. Organic farming seeks to build up and maintain high levels of soil organic matter, to increase the long-term fertility of soils, to work as much as possible within a closed cycle with regard to organic matter and nutrients and to minimise all forms of pollution (International Federation of Organic Agriculture Movements (IFOAM)). To achieve this, organic farmers minimise the use of inorganic fertilisers and avoid pesticides. Tillage techniques are usually minimised to allow soil organic matter to accumulate.

There are many non-greenhouse gas benefits of organic farming – the most obvious one being reduced use of pesticides and herbicides leading to reduced impacts on human health and wildlife, and reduced groundwater pollution. Many of the characteristics of organic farming may also have benefits in terms of greenhouse gas emissions -–such as a reduced demand for fuel for tillage operations and lower inputs of ago-chemicals such as pesticides. However, the adoption of organic farming is not predicated by the use of waste-derived compost, which may equally well play a role in conventional agriculture. Also, the availability of waste-derived compost does not influence the decision to switch to organic farming – compost is available from other sources (eg ‘green’ manure) and is not a limiting factor. Quantification of the greenhouse gas benefits of organic, as opposed to industrial agriculture is therefore part of the much bigger issue of the comparison of overall environmental burdens of each system. These issues are clearly outside the remit of the present study.

Nevertheless, it is instructive to consider where further indirect greenhouse gas benefits from adoption of organic farming could arise. This is illustrated in Table A5.51, which gives an example of energy input for the production of a crop of winter wheat. Adoption of organic farming techniques would be expected to reduce the overall energy requirements of many of the items listed, in addition to avoiding the use of mineral N fertiliser already considered in the analysis. However, the precise impacts are very hard to quantify because of large variations in farming practices and conditions. Savings due to reduced tillage, for example, could be at least partly offset by additional energy needed to transport and spread large quantities of compost – at 50 to 100 tonnes/ha. However, if compost were applied just as a surface mulch, much lower rates of carbon sequestration in soil would be likely to result as mentioned earlier, so reducing the overall greenhouse gas benefit of composting.

Table A5.51: Example of energy used in crop production (winter wheat – see reference 102).

Item	GJ/ha
Cultivation (seedbed preparation,, sowing, harvest and transport of crop	3.3
Machines (manufacture)	1.18
Seed	1.27
Herbicides	1.15
Fungicides	0.69
Total of above	**7.59**
N fertiliser* (120 kg / ha)	5.7
Grand total	**13.29**

*N fertiliser energy contribution based on 47 MJ/kg,, representative of energy use for production in the 1970s.

Reduction of N_2O emissions from denitrification in soils

A further indirect greenhouse gas impact of compost use arises from N_2O emissions from soils. Nitrous oxide is produced in soils mainly by the microbially-mediated processes of nitrification and denitrification [103], summarised in Box 10. There is a strong relationship between mineral N concentrations in the soil and emissions of N_2O, which applies whether the nitrogen originates from fertiliser or manure applications, mineralisation in the soil or atmospheric deposition. For inorganic fertilisers, it is estimated that between 1 to 3 % of the N applied is lost as N_2O.

Estimating impacts of compost use on N_2O release avoided due to substitution of mineral N fertilisers is subject to a great deal of uncertainty and has not been included in the present study. However, to investigate the possible impact of this factor on the greenhouse gas impacts of compost use, we will explore the use of a trial value of 2% of mineral N fertiliser being denitrified – a value mid range in the 1-3% estimated by Mosier [103]. If we allow the mean compost N content of 6.2 kg N / tonne fresh weight of compost to replace 30% of an equivalent quantity of mineral fertiliser N, and we assume that compost N does not give rise to N_2O emissions (because the N in compost is mineralised slowly), then this could avoid some 0.06 kg N_2O / tonne[a]. Given a GWP value for N_2O of 310 (over a 100-year horizon), this avoided emission would equate to ~20 kg CO_2 equivalents / tonne of compost. The impact of compost use on greenhouse impacts via the denitrification mechanism is therefore potentially large, but subject to a great deal of uncertainty, especially in terms of how compost compares with mineral fertiliser. In view of this uncertainty, we did not feel that inclusion of this effect in the analysis could be justified.

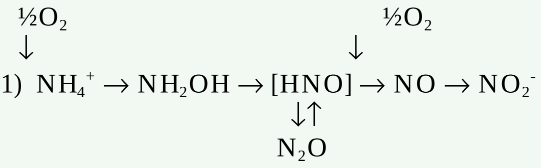
---

[a] 2% of 6.2 kg N / tonne of compost is 0.124 kg N as N_2O emission avoided / tonne compost. Given that there are 28 kg of N in 44 kg of N_2O, the avoided emission is equivalent to 0.195 kg N_2O avoided / tonne of compost.

Box 10: Nitrification and denitrification.

Nitrification is the oxidation of ammonium to nitrite and then nitrate. Some N_2O is formed from the breakdown of the intermediate hyponitrous acid, HNO. Nitrification requires O_2 from the air and is undertaken in the soil by several genera of bacteria, the most important of which are *Nitrosomonas*, *Nitrosococcus*, *Nitropira*, *Nitrosolobus* (ammonium to nitrite, equation 1) and *Nitrobacter* (nitrite to nitrate, equation 2).

Nitrification reactions:

(1)


$$\overset{\overset{\frac{1}{2}O_2}{\downarrow}}{NH_4^+} \to NH_2OH \to \underset{\underset{N_2O}{\downarrow\uparrow}}{[HNO]} \overset{\overset{\frac{1}{2}O_2}{\downarrow}}{\to} NO \to NO_2^-$$

$$(2)\ \ NO_2^- + \tfrac{1}{2}\,O_2 \to NO_3^-$$

Denitrification occurs under anaerobic conditions, either when the soil is waterlogged and air spaces are taken up with water, or in locally anaerobic micro-sites in otherwise aerobic soils. Denitrification is the reduction of nitrate and nitrite to N_2O and dinitrogen (N_2) (equation 3). Several bacterial genera can undertake denitrification reactions, including *Rhizobium*, *Pseudomonas*, *Bacillus* and *Paracoccus*.

Denitrification reaction:

$$(3)\ \ NO_3^- \to NO_2^- \to NO \to N_2O \to N_2 + H_2O$$

Because the rate of release of mineral N from compost takes place relatively slowly, uptake by crops can reduce the build up of substrates for denitrification, compared with the high concentrations that may result from the use of inorganic fertilisers or manures with a high content of available N. Other factors that influence denitrification rates are soil pH (rates increase up to about pH 8) and moisture content, with rates increasing after wetting.

OVERALL GREENHOUSE GAS FLUXES FROM COMPOSTING

Overall greenhouse gas fluxes from paper and putrescible waste are summarised in Table A5.52, which recapitulates data presented in the separate sections of this Appendix.

Table A5.52: Emission factors for wastes processed through composting (kg CO2 eq/t material treated).

Waste management option	**Waste component**	**Short cycle CO_2**	**Fossil CO_2**				**Short cycle C sequestered**	**Sum of fossil C and sequestered C**	**CH_4 emission**	**N_2O emission**	**Total GHG flux**
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	**(GWP=-1)**		GWP=21	GWP=310	
Open composting											
	Paper	**1144**	0	13	-36	8	-22	**-37**	0	0	**-37**
	Putrescible	**670**	0	13	-36	8	-22	**-37**	0	0	**-37**
Closed composting											
	Paper	**1144**	0	**18**	**-36**	**8**	**-22**	**-32**	0	0	**-32**
	Putrescible	**670**	0	**18**	**-36**	**8**	**-22**	**-32**	0	0	**-32**
Home composting											
	Paper	**1144**	0	0	**-36**	**0**	**-22**	**-58**	0	0	**-58**
	Putrescible	**670**	0	0	**-36**	**0**	**-22**	**-58**	0	0	**-58**

The greenhouse gas flux from composting the paper or putrescible waste in 1 tonne of MSW may be estimated by multiplying the above data by the proportion of paper and putrescibles in average MSW, as given in Table 9 on page 25.

COMPOSTING COSTS

Compost plants may offset their costs through sales of compost. The price of compost produced by centralised schemes varies widely across the EU. A limited amount of compost can fetch up to 40 Euro/t in specialised horticultural markets. However, as compost production increases more compost will go to agricultural markets, where it is either given away free or fetches prices of up to 15 Euro/t in northern European countries. In southern European countries the price could be as high as 50-100 Euro/t as the value of compost for improving the dry, arid, depleted soils which cause problems in these regions is particularly high[66]. The DHV report on composting104 concluded that if all of the organic waste collected was composted, there is sufficient agricultural land to provide a market for the compost in every member state except Luxembourg. However, revenue from compost sales is included within the gate fee and is therefore not accounted for separately in the model.

Gate fees

Not all composting plants charge gate fees – some recover some or all of their costs through standard waste management charges levied per household by the local waste management authority. Reported gate fees vary widely depending on the type of process and the cost of nearby landfill or incineration plants. Fees quoted in the RVF report [65] range from 37 to 42 Euro/t for plants in the Netherlands and 58 to 174 Euro/t for German plants. A survey of successful composting schemes across the EU quotes fees of 16 Euro/t (Spain), 22 Euro/t (Italy) and 41 Euro/t (Ireland). A report on plants in the Netherlands quotes fees from 46 to 70 Euro/t [105]. We have assumed a typical gate fee of 35 Euro/t for open plants and 50 Euro/t for closed plants.

Appendix 6. Anaerobic digestion (AD)

AD PROCESS DESCRIPTION

Anaerobic digestion involves the biological decomposition of waste in air-tight vessels under anaerobic conditions to produce a methane-rich biogas. The temperature, pH and moisture content are controlled to optimise methane production and the gas produced is collected and burnt for heat or electricity production. Anaerobic digestion is widely used in the water industry where it has been used for sewage treatment since the end of the 19th century. During the last two decades, systems have been developed for treating animal waste e.g. on farms, and for the putrescible component of MSW. The feedstock for AD can also contain up to 40% paper, although paper is not widely used at present. Here we have assumed that the feedstock is 10% paper and 90% food and garden waste. As for composting, source-separated waste is essential if the solid residue 'digestate' is to have value in agricultural or horticultural applications, as opposed to use as landfill cover or site restoration.

Incoming source-segregated waste is first screened and then mixed with previously digested material or liquor to inoculate it with bacteria and achieve the correct consistency. This mixture is then pumped in to the air-tight digester vessel where is held for 2-3 weeks. Whilst inside the digester the material is mixed and the biogas evolved (containing 55 – 65% methane by volume) is taken off and burnt for energy recovery. The solid residue extracted from the digester, termed digestate, is then de-watered in a screw press to achieve a moisture content less than 50% and the press water is returned for mixing with fresh feedstock.

A wide variety of designs of AD system are available, including single and multiple vessel designs. Control over temperature is very important since methane formation decreases markedly below about 30 deg C. Systems are available that operate up to 65 deg C, but although the methane yields tend to be higher and the process goes to completion faster, the higher temperature creates additional on-site energy demand for biogas and the process may be harder to control.

The liquid from the process is generally disposed of to sewer, although it could be used to manufacture a liquid fertiliser if the quality was high enough. The solid digestate is usually 'cured' by composting aerobically for one to two weeks to stabilise the waste, release free ammonia and to allow the moisture content to reduce through passive drying. This reduces odour and produces a compost-like material. If the composted digestate quality is sufficiently high, it can be used for agriculture, in which case the climate change impacts are considered to be the same as those described for aerobically-produced composts, described in Appendix 5. Compost from AD plants has suffered from a perception of inferior quality to some extent, due to poor performance of some of the early plants. However, if source-separated food and garden waste is the feedstock and the process is properly controlled, there is no reason why compost from AD plants should not be of equal quality to that from composting plants. For this study we assume that 100% of the compost is marketable.

AD GREENHOUSE GAS EMISSIONS

The following steps are considered in the analysis:

- **Mobilisation.** This includes collection and transport of waste from the householders to the AD plant and transport of products (composted digestate and liquor) to market or landfill.
- **Treatment.** Emissions of short-term carbon dioxide and leakage of methane during the AD process. More carbon is released as carbon dioxide during the composting of the digestate residue. This is all short term carbon dioxide as it originates from degradable organic matter. Energy use to operate the plant is provided by the AD gas.
- **Use / Disposal.** Carbon sequestered in soil as a result of composted digestate application.
- **Displaced emissions.** Avoided emissions from energy generation displaced by the heat and power exported by the AD plant. Also avoided emissions from displacement of peat or fertilisers by the composted digestate, as discussed in the section on composting.

Mobilisation

Emissions from mobilising waste for AD are considered to be the same as for composting. The relevant emission factors are shown in Appendix 5 (in Table A5.45, on page 139.

TREATMENT EMISSIONS

Anaerobic digestion (AD) produces a gas rich in methane – we assume 60% methane, the remainder being short-term carbon dioxide as it originates from degradable organic matter. Most of this methane is converted to short-term carbon dioxide during combustion of the gas for energy, but a little may escape through leakage.

In order to estimate the carbon released during the AD process for each of the two main waste streams which can be used: paper and putrescible waste, we initially followed the same approach as for composting and landfilling. This was based on an analysis of the carbon content of the waste, and estimates of the proportion of this carbon that will degrade. The fraction of DOC which dissimilates during anaerobic digestion (DDOC) could differ from that during landfilling for two main reasons:

- In AD, material is shredded and mixed in a liquid medium, and kept at optimum conditions of temperature to maximise the degradation process. This does not apply during landfilling, for the reasons discussed in Appendix 5 on composting.
- The AD process takes only a few weeks, whereas waste resides permanently in landfill sites.

Whilst the first points could imply higher dissimilable fractions for AD than for landfilling, the second issue could imply the opposite. In the absence of better data, we did not feel justified in selecting different DDOC values for AD than for landfilling. The estimates used for degradation during the AD process are therefore the same as for landfilling, and for aerobic composting. The results have already been presented in relation to composting in Table A5.46 in Appendix 5 (page 140).

A small amount of biogas may escape before combustion, resulting in the release of methane. We have found no data on fugitive methane leakage, which by definition is difficult to measure. If we were to assume that 0.5% of the methane formed subsequently escapes, then this would amount to some 0.46 kg CH_4 /tonne of putrescible waste, equivalent to about 10 kg CO_2 / tonne, assuming a global warming potential of methane of 21. It has been assumed in the analysis that no leakage of methane occurs. However, the above calculation indicates the considerable potential of even small leaks to impact on the overall greenhouse gas flux from AD.

USE / DISPOSAL AND DISPLACED EMISSIONS

Compost produced from AD digestate is treated in this analysis in the same way as compost produced by aerobic processes, as described in Appendix 5. AD compost is therefore credited with avoided emissions due to carbon sequestration and displacement of peat and fertilisers. However, in addition to these factors, the AD option also allows energy recovery from the biogas which is not available for aerobic composting. The displaced emissions associated with biogas recovery are described below.

The biogas is generally burnt in an engine, producing both electricity and heat. Around one third of the electricity produced through combustion of the gas is used internally for running the plant, and the rest is available for export. The heat may be recovered too, and again some of this is used internally and the surplus can be exported. This exported energy replaces heat or electricity generated elsewhere, with a consequent reduction in emissions. This reduction is dependent on the efficiency of the AD plant. Plants that export waste heat as well as electricity (i.e. CHP plants) achieve a much greater reduction in emissions. AD plants both with and without CHP have been evaluated here. Alternatively the gas itself can be burnt directly as a fuel in an adjacent industrial plant such as a brickworks, or it can be converted to a transport fuel. Biogas is used successfully to power waste collection vehicles in Sweden. However, these applications are less common and have not been evaluated here.

The assumptions used in calculating energy production from AD plants are shown in Table A6.53. They are partly based on an analysis of figures reported for real AD plants in various EU countries [65]. The average of the reported figures for the proportion of methane in biogas was 62% but we rounded this to 60%. Heat and electricity generation, own use and export vary widely between the plants so we selected 'typical' figures to represent an average plant.

Table A6.53 Parameters for AD energy production

Parameter	Value
Proportion of methane in biogas	60%
Efficiency of electricity generation	30%
Proportion of electricity used in-house	33%
Proportion of waste heat collected for CHP option	80%
Proportion of heat used in-house for CHP option	15%
Calorific value of methane	10 kWh/m^3
Proportion of methane leaking	0.5%

Using the assumptions described above for degradation of putrescible waste and paper, we estimate that in theory around 260 kg of biogas should be generated per tonne of putrescible waste or paper. However, this figure is twice that observed in practice: typically around $100m^3$

(130 kg) of biogas per tonne of waste. For example, the average of five schemes reported in the RVF report [65] ranging from 110 to 190 kg/t was 140 kg/t (110 m^3/t) putrescible waste.

This anomaly arises because of three reasons:

1. Some carbon is lost as CO_2 during degradation of waste prior to arrival at the AD plant.
2. Some carbon is lost as CO_2 dissolved in the AD liquor. Initially the waste generates approximately equal proportions of CO_2 and methane (for paper waste, a little more methane for food and garden waste), but around half the CO_2 dissolves in the liquor.
3. Some carbon is incorporated into organic compounds in the AD liquor.

For these reasons, we have used the reported figures for biogas generation in the estimate of avoided energy instead of our calculated theoretical figures for each material. This precludes a material-specific breakdown into paper or putrescible waste.

The results are shown in Table A6.54. The avoided emissions were calculated using the average electricity and heat generation mixes for the EU.

Table A6.54: Avoided emissions from electricity and heat generated from AD

Parameter	Value
Biogas yield	100 m^3 /t waste
Calorific value of biogas (60% CH_4)	600 kWh/t waste
Electricity generated (30% efficiency)	180 kWh/t waste
Electricity for export (70% of elec. gen)	120 kWh/t waste
Avoided emissions from electricity export	54 kg CO_2/t waste
Heat recovered for CHP option (80%)	336 kWh/t waste
Heat exported for CHP option (85%)	286 kWh/t waste
Avoided emissions from CHP heat export	81 kg CO_2/t waste

OVERALL GREENHOUSE GAS FLUXES FROM AD

Overall greenhouse gas fluxes from paper and putrescible wastes are summarised in Table A6.55, which recapitulates data presented in the separate sections of this Appendix.

Table A6.55: Emission factors for wastes processed through AD (kg CO2 eq/t material treated).

Waste management option	**Waste component**	**Short cycle CO_2**	**Fossil CO_2**				**Short cycle C sequestered**	**Sum of fossil C and sequestered C**	**CH_4 emission**	**N_2O emission**	**Total GHG flux**
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	**(GWP=-1)**		GWP=21	GWP=310	
AD – energy recovery as electricity only											
EU-average electricity	Paper	**721**	0	0	-90	8	-22	**-104**	0	0	**-104**
	Putrescible	**250**	0	0	-90	8	-22	**-104**	0	0	**-104**
Wind power	Paper	**721**	0	0	-37	8	-22	**-51**	0	0	**-51**
	Putrescible	**250**	0	0	-37	8	-22	**-51**	0	0	**-51**
Coal power	Paper	**721**	0	0	-151	8	-22	**-165**	0	0	**-165**
	Putrescible	**250**	0	0	-151	8	-22	**-165**	0	0	**-165**
AD – CHP energy recovery											
EU-average electricity	Paper	**721**	0	0	-171	8	-22	**-185**	0	0	**-185**
	Putrescible	**250**	0	0	-171	8	-22	**-185**	0	0	**-185**
Wind power	Paper	**721**	0	0	-118	8	-22	**-132**	0	0	**-132**
	Putrescible	**250**	0	0	-118	8	-22	**-132**	0	0	**-132**
Coal power	Paper	**721**	0	0	-232	8	-22	**-246**	0	0	**-246**
	Putrescible	**250**	0	0	-232	8	-22	**-246**	0	0	**-246**

The greenhouse gas flux from digesting the paper or putrescible waste in 1 tonne of MSW may be estimated by multiplying the above data by the proportion of paper and putrescibles in average MSW, as given in Table 9 on page 25.

AD COSTS

The RVF report quotes gate fees of 41Euro/t (France), 42Euro/t (Finland), 50Euro/t (Sweden), 62Euro/t (Belgium) and 153Euro/t (Germany). Each of these figures is for just one plant and cannot necessarily be deemed representative. A report on plants in the Netherlands reports gate fees from 76 to 82 Euro/t[105]. The figures average to 70Euro/t, but as this is skewed by the very high figure for Germany we have reduced this to 65Euro/t for our analysis.

Appendix 7. Recycling

Recycling of materials from the municipal solid waste stream generally involves the following steps:

1. Collecting the separated materials from individual households and transporting to a place for further treatment
2. Sorting, baling and bulking for onward transfer to reprocessors (e.g. at a Materials Recycling Facility [MRF])
3. Reprocessing to produce marketable materials and products

The costs of reprocessing depend largely on the material to be processed, the scale of the process used, the complexity of the reprocessing technology, and the quality of the input materials.

The following sections describe typical routes taken by various materials and waste products from the household in order that they may be recycled and returned to beneficial use. It is possible that innovative and cost-effective alternative ways of recycling could be developed in the future, but it is difficult to predict where these are likely to occur. In addition, legislative pressures to achieve set targets for recycling and recovery may lead to shifts in the balance of waste management treatment options as certain recycling routes become more cost-competitive with respect to other routes.

WASTE SORTING

Sorting Process Description

A typical Materials Recycling Facility (MRF) operation employs manual processing and/or semi-automatic processing for sorting recyclables from waste and preparing them in a form suitable for use by a materials reprocessor. A MRF is typically equipped with conveyors, trommel (rotary screen) separators (for removal of fine particles and permitting size separation) picking lines where manual operators remove recyclable materials, and baling equipment. Other equipment includes weigh-bridges, storage containers, fork-lift trucks, and some MRFs may be equipped with magnetic separators, eddy current separators and machines for the detection and separation of plastics by polymer type. There are various options for the equipment layout of a MRF depending on the degree of source-separation already performed by the householder. Generally, well separated recyclables require less processing at a MRF than unsorted, mixed recyclables.

The model assumes a ‘clean’ MRF receiving source-separated recyclable materials. ‘Dirty MRFs’, which receive unsegregated dustbin waste, are widely used in the USA, where sink disposal units are widely used for getting rid of kitchen wastes which would otherwise end up in the household waste stream. Dirty MRFs have not achieved widespread deployment in Europe because of the higher content of putrescible material in the waste stream.

Greenhouse Gas Emissions from sorting

Emissions arise from transport of materials to the MRF and from the MRF to the reprocessor, and from energy use at the MRF.

Transport

There are three possible collection routes:

- 'bring' recycling i.e. delivery of recyclables by householders to collection points at supermarkets etc;
- delivery to a household waste site;
- kerbside collection.

It is assumed that 20% of the material is collected via household waste sites and 40% by each of the other routes. It is assumed that no emissions arise from the delivery of recyclables to collection points for 'bring' recycling as special trips are not made. However, emissions from collection of the waste from the collection points and delivery to the MRF are included.

Waste electrical and electronic equipment (WEEE) may follow a different route – it can be either taken back by the retailer or taken to a specialised collection point. Take-back by the retailer would generally not require a separate trip. However, much WEEE (such as old 'white goods') may be taken to household waste sites. It is assumed that 60% of WEEE is taken to household waste sites or specialised collection points, and 40% collected via take-back.

Transport routes:

kerbside
Household → Materials recovery facility (MRF)→ reprocessor
household waste site
Household → Household waste site → Materials recovery facility (MRF)→ reprocessor
bring
Household → Collection point→ Materials recovery facility (MRF)→ reprocessor

Vehicle emission characteristics and assumed journey distances are given Appendix 1. Emissions are summarised in Table A7.56.

Table A7.56 Mobilisation emissions associated with recycling

From	To	Load	Vehicle	% by route	CO_2 emissions kg/t
Collection point	MRF	Mixed recyclables	L1	40%	1.4
Household	HHW site	Mixed recyclables	car	20%	2.6
HHW site	MRF	Mixed recyclables	L1	20%	0.7
Household	MRF	Mixed recyclables	L1	40%	1.4
Household	MRF	Plastics	L1	40%	7.1
Household	retailer	WEEE	car	40%	0.0
Household	HHW site	WEEE	car	60%	7.7

Table A7.56 continued

From	To	Load	Vehicle	% by route	CO_2 emissions kg/t
MRF	Landfill	Residue	L1	100%	0.2
MRF	Paper reprocessor	Paper	L2	100%	4.0
MRF	Glass reprocessor	Glass cullet	L2	100%	4.0
MRF	Ferrous metal reprocessor	Ferrous scrap	L2	100%	4.0
MRF	Aluminium reprocessor	Aluminium scrap	L2	100%	4.0
MRF	Plastic reprocessor	Plastic feedstock	L2	100%	4.7
Collection point	Textile reprocessor	Textiles	L2	100%	4.0
Collection point	WEEE reprocessor	WEEE	L2	100%	4.2
Total	Paper				**10.31**
Total	Glass				**10.31**
Total	Ferrous metal				**10.31**
Total	Aluminium				**10.31**
Total	Plastic				**15.29**
Total	Textiles				**10.31**
Total	WEEE				**12.08**

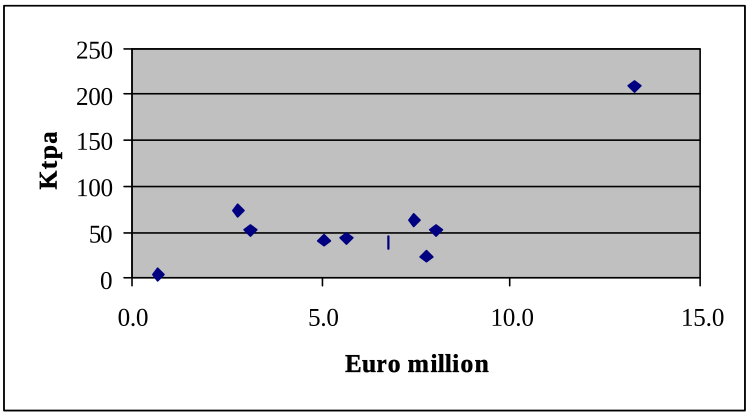
Energy use

A standard figure of 25 kWh of electricity per tonne of material sorted is used. This was taken from a recent UK study [106] and is similar to the figure of 19kWh/t waste plus one litre of diesel quoted in [107].

MRF COSTS

The capital cost of a MRF depends on its size and complexity, and costs can range from about Euro7M for a 40-50ktpa plant (Euro140k/tpa to 175 k/tpa) to about Euro14 million for 200ktpa capacity (Euro70k/tpa) (Figure A7.40).

Figure A7.40: Quoted Capital Costs for Various MRF Capacities



Total costs per tonne of waste processed at a MRF (i.e. including capital and operating costs) have been estimated as Euro51/tonne and Euro128/tonne respectively for a manual processing and semi-automatic processing plant[107] . These costs are roughly in line with costs determined in a USA study by the National Solid Waste Management Association (NSWMA) carried out in 1992, which found an average cost of Euro45/tonne[108.] The study also derived costs per tonne of material (Table A7.57). Care must be taken in interpreting these cost/ton figures, as to date MRFs tend to concentrate on processing paper and cardboard, and consequently the processing costs/tonne are kept low. High cost/tonne figures tend to be associated with materials processed at much lower volumes.

Table A7.57: Processing Costs at MRF by Material

Material	Cost Euro$_{1999}$/t
Newspaper	31
Corrugated	40
Mixed paper	34
Aluminium cans	180
Steel cans	85
Clear glass	67
Brown glass	103
Green glass	80
Mixed glass	46
PET plastic	169
HDPE plastic	172

An average cost of Euro51/tonne of waste processed at the MRF (i.e. after reference [107], assuming manual processing) is adopted in the study; this cost is not allocated differentially across materials.

PAPER

Paper Recycling Process Description

Typical routes for the collection and reprocessing of wastepaper are illustrated in Figure A7.41. Paper and board constitutes a significant proportion of household waste (typically one third). The collection of wastepaper from households currently tends to be focused on the collection of old newspapers and magazines, which make up about 10% of the municipal waste stream. This is mainly because these items are less likely to be contaminated (e.g. with food residues). However, household collections of old cardboard are also carried out.

Paper and board consumption tends to increase in proportion to an increase in gross domestic product (GDP), and overall production in the EU has risen from 60 million tonnes in 1993 to over 70 million tonnes in 1997. EU production was about 6 million tonnes higher than consumption in 1997, and overall the EU is a small net exporter of paper and board. Production and consumption are not evenly balanced in all Member States though, with Austria, Finland and Sweden producing considerably more paper than they consume, and this is generally due to the availability of raw materials for paper production (particularly in Finland and Sweden).

The availability of paper and board for recycling is conditioned by the internal consumption of paper and board, although not all papers and boards can be collected for several reasons including:

- Destruction in use (e.g. household and sanitary papers);
- Contamination (e.g. paper and board from refuse sorting stations is unsuitable for paper recycling);
- Immobilisation (e.g. books), and
- Wide dispersion of some sources (e.g. villages, low population areas).

Figure A7.41: Typical Routes for Waste Paper Recycling

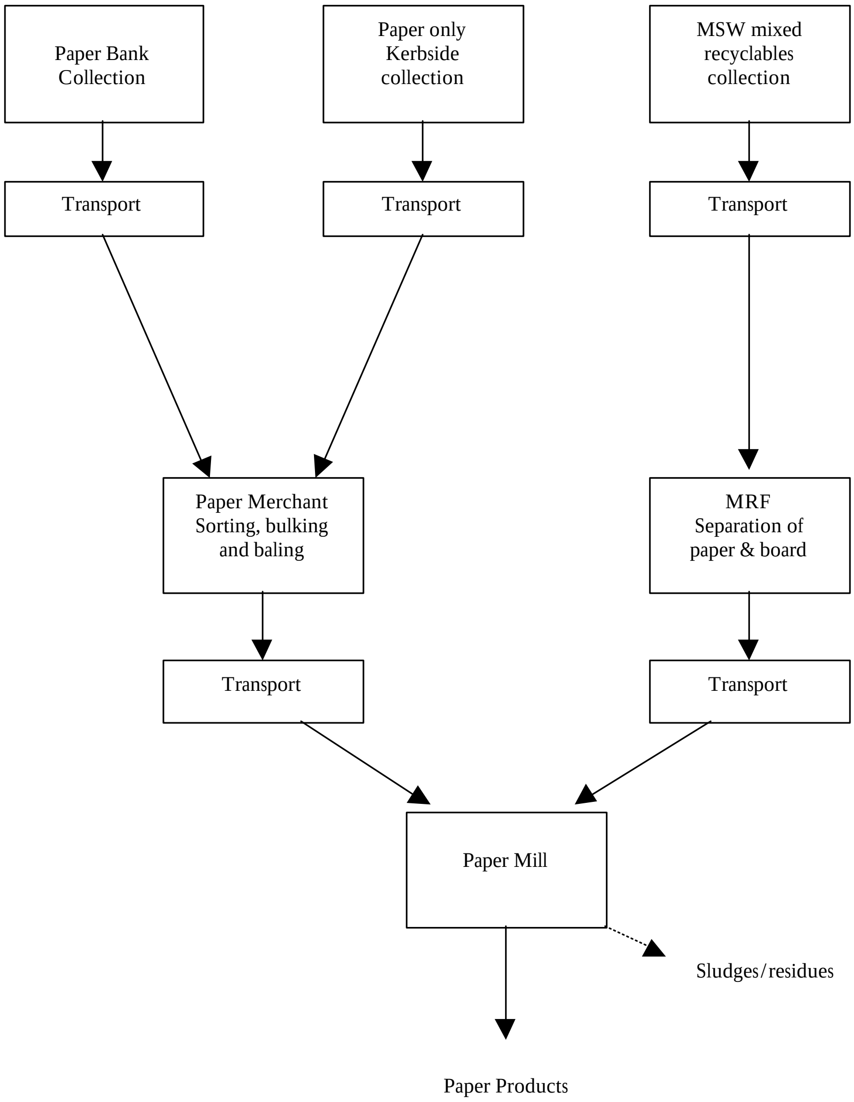


Table A7.58 shows the fraction of paper consumption which was collected for recycling in Member States in 1998. Generally, the industry has estimated that more than 30% of the theoretical source of paper and board (i.e. national consumption) cannot practically be collected. Some Member States (e.g. Germany, Austria and Sweden) are already close to this limit.

Table A7.58 Collection of Paper for Recycling in 1998 (as percentage of paper consumption)

Country	Collection rate (%)
Austria	65
Belgium	48
Denmark	52
Finland	63
France	41
Germany	70
Greece	21
Ireland	21
Italy	33
Netherlands	58
Portugal	42
Spain	43
Sweden	64
UK	40
EU	49

Note: Luxembourg not included in original source data
Source: CEPI, 1999.

The paper industry comprises a number of discreet manufacturing sectors, each producing paper grades to meet the requirements of a specific market. There is potential to use waste paper in the manufacture of most commodity grades. However, the type of waste paper used as feedstock, and the plant required to recycle it, will vary enormously between sectors.

In general the manufacture of lower grade paper products such as corrugated case will tolerate a low quality feedstock, such as container waste. Similarly the recycling plant required to produce papermaking stock for low grade paper will involve only a limited number of process stages. Manufacture of fine paper grades paper will, by contrast, require a high quality feedstock comprising predominantly of wood free (office) waste and will require a complex process comprising of multiple stages of de-inking, screening and cleaning to produce a stock of comparable quality to virgin pulp.

Paper Recycling Greenhouse Gas Emissions

Several sources of data on greenhouse gas emissions associated with paper production from virgin material and recycled paper have been considered.

1) Data on newspaper has been taken from the US EPA report [67] which examined greenhouse gas emissions associated with waste management.
2) This was compared with data on heat and electrical energy consumption from a 1996 Swedish study reported in an EEA report[109] which we converted into CO_2 emissions using EU average emission factors for heat and electricity production.
3) We also have data from the BUWAL life cycle database for the production of various paper types. We have compared figures for the production of unbleached kraft (packaging) paper from wood with the production of Swiss kraft paper from paper containing 83% recycled content. Both sets of figures are for Swiss production plants and therefore are based on the Swiss energy mix. We have also compared figures for graphic paper and board. However it is important to note that these processes are from different plants and do not produce directly equivalent products and therefore comparison should be made with care.

4) BREF notes on pulp and paper making give some figures on emissions from stock preparation for recycled paper but these do not include the subsequent paper-making stage.

The figures are shown in Table A7.59.

Table A7.59 Life Cycle Emissions for Production of paper (kg CO_2/t paper)

paper type	source	from virgin materials	from recycled materials
newsprint	Swedish study	1755	849
newsprint	US study	2222	1535
newsprint	BUWAL database		291 (68% recycled)
kraft paper unbleached	BUWAL database	1080	633 (Swiss kraft)
graphic paper	BUWAL database	436 (uncoated) 730 (coated)	586 with deinking 380 without deinking
corrugated board	BUWAL database	644 (25% recycled)	522-556
packaging paper	BREF note		90 (stock only)
newsprint	BREF note		215 (stock only)

It can be seen that the range of data is large. For this study we took the Swedish figures because they were based on the EU average energy mix. However it should be noted that these figures are only for newsprint recycling. The figures from the BUWAL database suggest that CO_2 savings may be smaller for other paper grades.

It is assumed that 0.7 tonnes of newsprint is produced from 1 tonne of recycled newspapers.

Paper Recycling Costs

The average European price paid for newsprint delivered to paper mills is 475Euro/t.
Higher grades will command higher prices. This are broadly in line with costs reported by the British Newsprint Manufacturers Association110, which gave unit costs of new paper mills producing newsprint from de-inked pulp across Europe, as in the range Euro410-470 per tonne, 46 % of which was operating costs (40% capital charges and 14% raw material costs).

GLASS

Glass Recycling Process Description

Glass is typically collected either via a bottle bank bring system or kerbside collection and taken to a glass merchant. From here it goes to a glass manufacturer, where the cullet (crushed glass) is used as a raw material in the production of new glass. Cullet aids the melting process and permits the glass furnace to be operated at lower temperatures than would be necessary for melting raw materials alone, allowing fuel savings to be made. Almost all glass production already involves the use of some cullet.

There are a large number of potential alternative uses for waste container glass. These include the use of cullet in the manufacture of construction materials like bricks, foam building blocks, cement, ceramic tiles, clay pipes and decorative products. In brick production, for example, there would be an added benefit of a reduction in kiln firing temperatures and fuel consumption. However, the costs for grinding the glass to a fine particle size do not make this process economic at current fuel prices. There is also potential for cullet to be used as aggregate substitutes. The main ‘constraint’ to alternative usage of glass cullet until now has been the usually superior value of, and generally high demand for, recycling cullet back into glass containers (bottles and jars).

Although the technical problems of alternative usage of waste glass can be overcome, the main barrier to its use in these applications is cost as:

- cullet has to compete with low-cost aggregates, hence the price of cullet needs to be low
- the cost of grinding cullet to the particle sizes required by certain applications (e.g. asphalt, brickmaking, or cement manufacture) is often comparable to the cost of traditional raw material.

This study therefore models only the conventional use of recycled glass in container manufacture. The routes involved are shown in Figure 4.4.

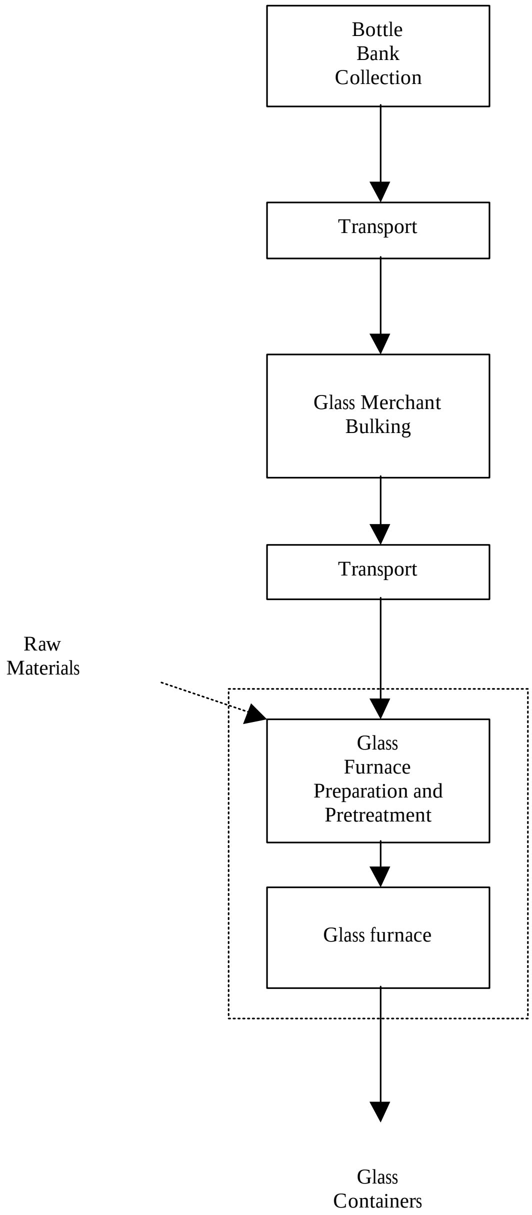
EU glass container production was about 17.4 Mt in 1997, but the EU is a large net exporter of glass via filled containers, which is why estimates of glass container waste arisings are significantly lower (around 14 Mtpa). Recycling rates vary considerable between Member States (Table A7.60).

Table A7.60: Glass Recycling Rates (1997)

Country	kt collected	recycling rate (%)
Austria	216	88
Belgium	222	75
Denmark	126	70
Finland	32	62
France	1,500	52
Germany	2,737	79
Greece	40	26
Ireland	36	38
Italy	750	34
Netherlands	375	82
Portugal	117	44
Spain	521	37
Sweden	134	76
United Kingdom	441	23

(Source: FEVE)

Figure A7.42 : Routes involved in glass recycling



Glass Recycling Greenhouse Gas Emissions

When calculating the greenhouse gas savings through the use of recycled glass, it is necessary to take account of the fact that current glass production already includes a significant use of recycled glass. We need to establish the additional emissions saved through the use of a further tonne of recycled glass.

To do this we have compared life-cycle data on the manufacture of high-cullet glass and low-cullet glass taken from the EA/Chem Systems life cycle inventory [114]. This data includes emissions from transport and manufacture of all raw materials as well as the energy used in glass manufacture. The data is shown in Table A7.61.

Table A7.61: Energy used to manufacture high cullet and low cullet glass

Input		Low cullet glass	High cullet glass
cullet		25%	59%
diesel in transport	MJ	514	282
grid electricity	MJ	411	469
energy unspecified	MJ	1226	672
coke oven gas	MJ	0	327
oil	MJ	1957	828
natural gas	MJ	2872	3265

From this data we have estimated that the carbon dioxide savings through the use of an additional tonne of cullet are 301 kg CO_2. 1049 tonnes of raw cullet are needed for 1000 tonnes of processsed cullet.

Glass Recycling Costs

Current prices paid by the reprocessor for collected cullet are shown in Table A7.62. A typical price of 34 Euro/t is used in this study.

Table A7.62 - Current Prices Paid for Collected Cullet

Colour Separation	Price Range Paid (Euro/tonne)
Green	0-17
Brown	25-37
Clear	37-45

(Source: Materials Recycling Week, May 2000)

ALUMINIUM & FERROUS METAL

Metal Recycling Process Description

Typical recycling routes are shown in Figure A7.43.

Aluminium Recycling

The main sources of aluminium metal in municipal solid waste are drinks cans and aluminium foil and foil containers. Cans and foils are different alloys of aluminium requiring different treatment processes, and where possible, it is desirable to ensure that cans and foils are recovered separately. Aluminium is also contained in consumer products such as domestic appliances, vehicles, kitchenware, toys etc., and these will usually find their way into the scrap collection chain via household waste sites, merchants and other local authority initiatives. The typical levels of aluminium present in municipal solid waste are quite low (~0.5%), but because of the very high intrinsic value of aluminium, recovery is viable.

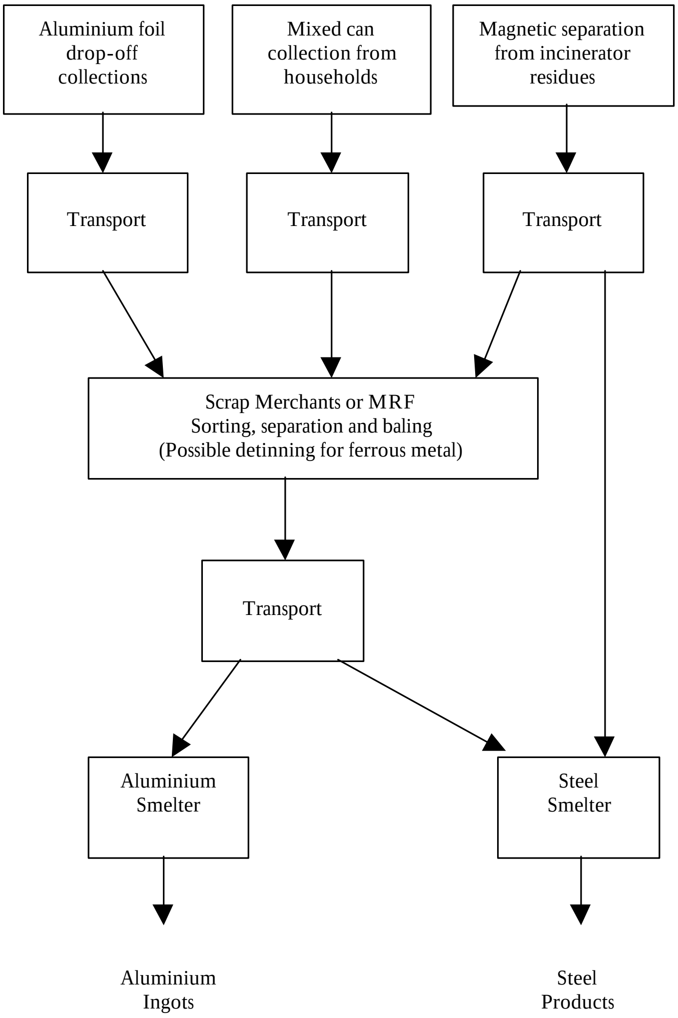
Aluminium foil recovery via bring systems helps householders to make the distinction between cans and foils and ensures that they direct them to the appropriate recovery channel. Magnetic and eddy current separation techniques can be employed to effect separation of ferrous metal from aluminium in mixed can collections and in incinerator residues.

Recycled aluminium is an integrated and increasingly important factor in meeting demand for 'new' aluminium products. 'Closed-loop' recycling is used where it makes sense, but the efficiency and economic well-being of the aluminium industry relies on the flexibility to be able to use post-use aluminium for production of different products at its discretion. Thus 'closed-loop recycling tends to make practical and economic sense for drinks cans whereas foil may be recycled into aluminium products for the automotive industry (e.g. high quality pistons).

Ferrous Metal Recycling

The reprocessing of scrap ferrous metal is a well-established industry. For example, in the UK, iron and steel recycling accounts for around 45% of the total input material to the iron and steel industry and large quantities of ferrous scrap are exported for recycling. Households are a significant but relatively very minor source of ferrous scrap in the form of tin-plated food cans. Magnetic separation is a commonly used operation to recover ferrous cans from the municipal waste stream.

Figure A7.43: Aluminium and Ferrous Metal Recycling



Metal Recycling Greenhouse Gas Emissions

Closed loop recycling of aluminium cans and steel cans is modelled (i.e. recycling of cans into more cans). Data on the greenhouse gas emissions for production of aluminium ingots from raw material and from recycled aluminium, and for production of tin plate from raw materials and from non-detinned scrap have been drawn from the BUWAL 250 data set[111]. This data includes all emissions associated with transport of materials, energy used in processes etc. For primary aluminium production, emissions of the potent greenhouse gas carbon tetrafluoride (CF_4), which has a global warming potential of 6500, are included. It is assumed that 0.93 tonnes of aluminium are produced from 1 tonne of recycled cans, and 0.84 tonnes of tinplate from 1 tonne of scrap.

Table A7.63: Greenhouse gas emissions for production of virgin and recycled aluminium and steel

Material	CO_2 (kg)	CF_4 (kg)	Total CO_2 eq (kg)
1000 kg aluminium ingot (virgin)	7640	0.4	10240
1000 kg aluminium ingot (recycled)	403	0	403
1000 kg tin plate (virgin)	2970		2970
1000 kg tin plate from non-detinned scrap	1160		1160

Metal Recycling Costs

Aluminium

The average price paid for used beverage cans in the EU is Euro945/tonne [112]

Ferrous Metal

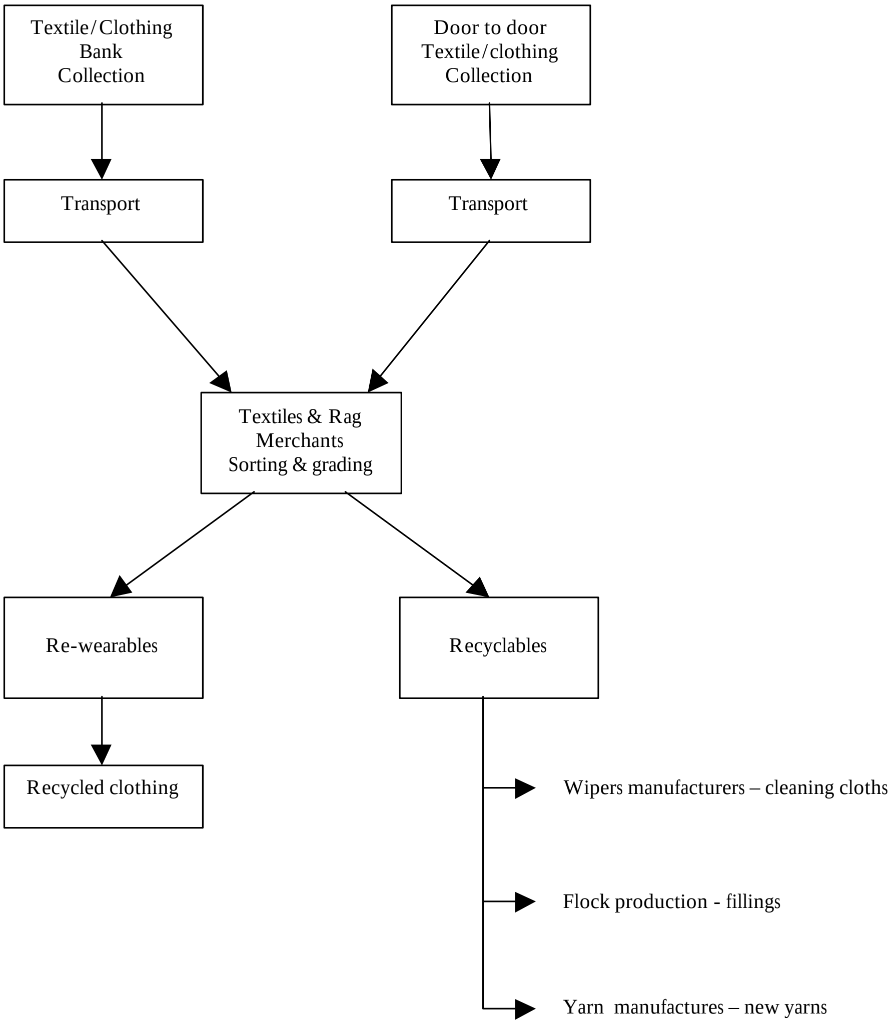
The average price paid for used steel scrap in the EU is Euro22/tonne [112].

TEXTILES

Textiles Recycling Process Description

Textiles available for recycling include old clothing, curtains and other household textiles and carpets. Textile banks (sometimes run by charities) may collect them, or they may be collected via kerbside collection. There are a number of potential recycling routes: rewearable clothes may be exported, clothes unsuitable for this and other textiles may be cut up to provide cleaning cloths (wiper production), used to produce flock fillings (e.g. for soft furnishings) or processed to produce new yarns (Figure A7.44).

Figure A7.44: Textile Reprocessing Routes



Data on textiles waste arisings and recycling is sparse; data for three countries is shown in Table A7.64.

Table A7.64: Recycling Routes for Textiles

	UK	Germany	Italy
Textile waste arisings	650 ktpa	2000ktpa*	
Collected	200 ktpa	500-700 ktpa (excl. carpets)	200 ktpa
Reuse (exports)	20%	40%	30%
Production of wipers	30%	25%	20%
Other recycling, e.g. new yarn, carpets, sound proofing	45%	25%	45%
Residue	5%	10%	10%

* 50% old clothing, 25% carpets, and 25% other materials, such as curtains and technical textiles

Textiles Recycling Greenhouse Gas Emissions

The recycling of textiles is represented in the model by recycling of textile fibres into wool and wool/acrylic garments. This leads to energy savings by avoiding the need for raw wool scouring, removal of contaminants from wool and dyeing; there is some additional energy usage from rag pulling. Overall energy savings are estimated to be 54 GJ/t, and are mainly in the form of gas giving a CO_2 saving of 3031 kg/t[113].

Textiles Recycling Costs

Data from a UK textile processing plant quotes the price paid for waste textiles as Euro1180/t [113].

PLASTICS

Plastics Recycling Process Description

Possible recycling routes for plastics are shown in Figure A7.45. Post-consumer plastics wastes (usually plastic bottles rather than plastic films) can be collected from households via a bring system or segregated kerbside collection. The density of loose quantities of plastics is very low (typically 25 to 30 $kg.m^{-3}$), and hence the payloads possible on collection lorries are very low (e.g. a lorry with a $60m^3$ carrying volume could only carry between 1 and 2 tonnes of loose plastics).

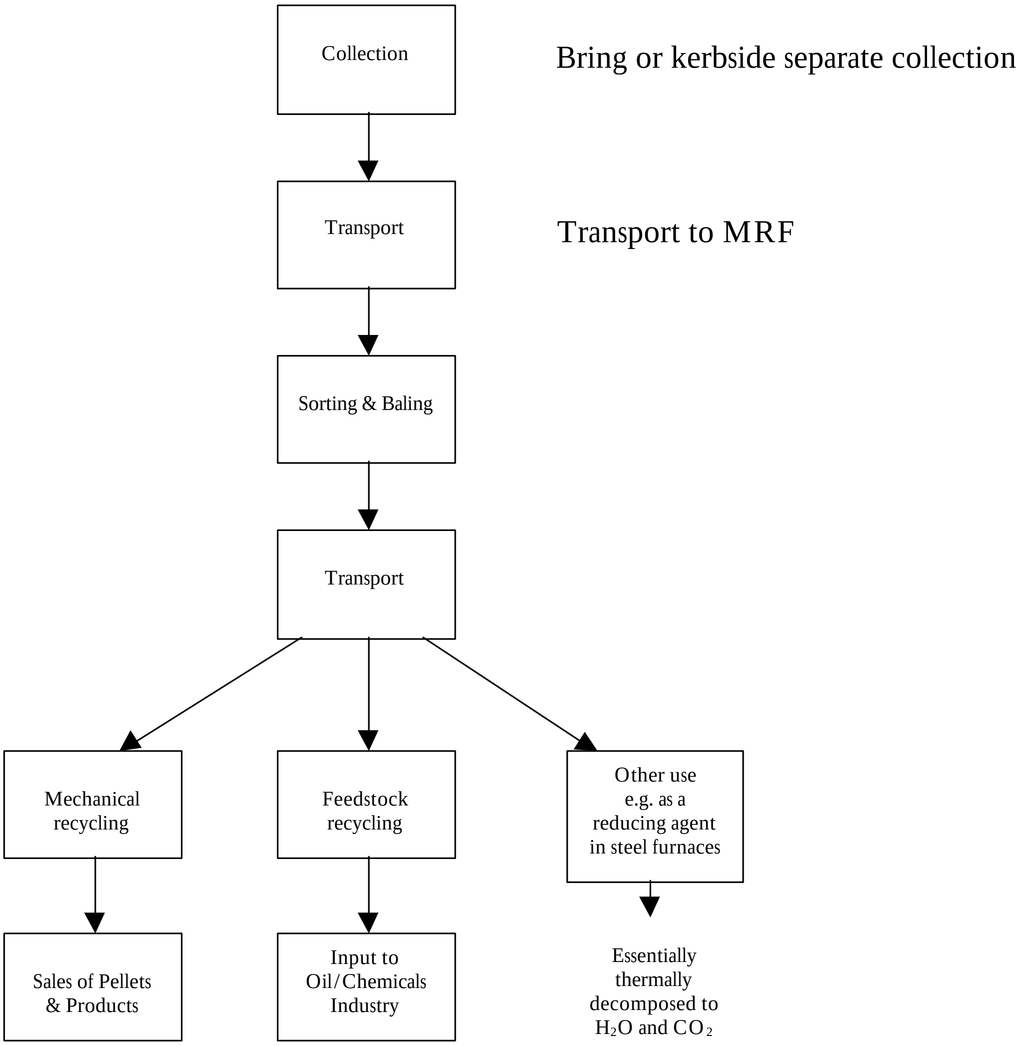
Plastic types contained in MSW are typically comprised of:

- High density polyethylene (HDPE):containers
- Low density polyethylene (LDPE): bags, toys, coatings, containers
- Polyethylene terephthalate (PET): bottles, textile fibres, film food packaging
- Polypropylene (PP): Film, microwave-proof containers, crates, electrical components
- Polystyrene (PS): tape cassettes, cups, plates, toys
- Polyvinyl chloride (PVC):bottles, cling film, toys, credit cards,
- Polyamide (PA): films for packaging foods such as oil, cheese and boil-in-bag products, textile fibres

On arrival at the MRF, the segregated plastic bottles are sorted according to polymer type. The sorting process can be either by hand or by automatic techniques or by a combination of both. Where possible coloured plastics are removed to form a low grade fraction (often referred to as ‘Jazz’). Sorted, single polymer bottles are then compacted and baled for onward transportation to plastic reprocessors. The density of the plastic bales is usually in the range 200 to 300 $kg.m^{-3}$ and payloads of 16-18 tonnes of baled plastics are typical.

Three types of reprocessing are possible – mechanical recycling, feedstock recycling and use as reducing agent.

Figure A7.45: Plastics Recycling Routes



Mechanical recycling:

In mechanical recycling, the plastic will be shredded or crumbled to a flake or granule. Cyclone separators may be used to remove contaminants like paper fibres from paper labels. Unless the plastic is known to be clean, the shredded material is washed to remove traces of food and soluble or dispersible dirt. At this stage settling or flotation techniques or hydrocycloning can be used to separate out other polymer types. The flakes/granules are then dewatered and dried and stored for onward sale to plastics compounders, or compounded in situ to produce the plastic end product in a pellet form suitable for sale or use for blow moulding, extrusion etc.

High quality recyclate, which can be used to substitute for virgin materials (either in the same product such as bottles, or a different product such as fibres) requires a very low degree of contamination. For lower quality recyclates which are heavily contaminated, (or have been developed from mixed plastics), the recyclate produced can be used to make a number of products, including plastic fencing, traffic cones, plant pots and industrial flooring.

Feedstock Recycling

This generic term is used to cover various processes (including depolymerisation processes, hydrogenation processes, alcoholysis processes, thermal cracking etc.) in which polymers are thermally disrupted to produce a hydrocarbon feedstock for the petrochemical industry. In depolymerisation, the polymer chains are broken down into smaller units (perhaps as far as the original monomeric constituents) which are subsequently refined and recombined to make new polymeric materials. Under thermal cracking, the breakdown chemicals produced are usually in a form suitable for input to conventional oil and petrochemical refineries. For example, the cracking process developed by BP Chemicals utilises a basic oxide fluidised bed reactor for the thermal cracking process. Acids produced during the cracking process (e.g. HCl from the breakdown of PVC) are neutralised in the basic conditions present in the bed, and contaminant metals are also deposited on the bed sand.

Use as a Reducing Agent

This involves using plastics as ‘de-oxidant’ (or reducing agent) in blast furnaces as a substitute for coal and/or fuel oil. The plastics are thermally degraded into CO_2 and water. One advantage of using plastic waste is the low sulphur content compared with coal. However, concerns have been expressed about the possible formation of dioxins and furans arising from the chlorine present in PVC – although measurements taken during experiments have proven that these are not formed in the strongly reducing atmosphere at 2,100°C, thus making chlorine removal pre-treatment unnecessary. Chlorine has no added value in the process and may only contribute to problems like corrosion in the blast furnace.

In 1998 in Western Europe, of 17.57 Mt of available collectable post user plastics waste (11.37Mt in MSW and 6.2Mt other sources), about 1.6 Mt goes for mechanical recycling and 0.4 Mt for feedstock recycling (Table A7.65).

Table A7.65: Plastics Waste Management in Western Europe 1998

Route:	ktpa
Mechanical recycling	1,576
Feedstock recycling	361
Energy recovery	3,348
Export for recycling	38

Source: APME

Plastics Recycling Greenhouse Gas Emissions

Data on the emissions associated with plastics production are available from the BUWAL 250 LCA data set[111] which is based on data from APME, except for HDPE where we used data from EA/Chem Systems. Data on recycling of HDPE plastic bottles into flakes which are then extruded into pellets which can substitute for virgin material is available for a plant in the UK[114],and gives a value of 341 kg CO_2/t recyclate due to a much lower energy demand. Similarly data on PET bottle recycling to produce PET flakes at a Swiss plant gives a value of 114 kg CO_2/t of flakes due to a low energy demand. We have compared this with data from the US EPA study[67]. Life cycle CO_2 emissions associated with the production of different types of plastics are given in Table A7.66

Table A7.66: Life Cycle Emissions Associated with Plastics Production kg CO_2/t

Plastic type	EU virgin	EUrecycled	US virgin	US recycled
PE Granules (general)	2200			
HDPE granules	1000	341	700	280
LDPE granules	2320		890	330
LLDPE granules	1910			
PVC powder	1940			
PET granules	2200	114	1160	450
PP granules	1800			

For this study we have taken the EU-based estimates for HDPE and PET, although the CO_2 savings appear generous when compared with the US data.

Plastics Recycling Costs

Figures from RECOUP[115], the UK organisation specialising in plastic container recycling give the total costs of mechanical recycling in the range Euro845 to 915 per tonne. Given that typical revenues for recycled plastics are in the range Euro560 to 635 per tonne, it is clear that economics is not driving current recycling activity and that some subsidisation is occurring.

These figures compare well with earlier reported figures[116] for HDPE mechanical recycling, where the total cost of recycled HDPE was put at Euro775/tonne (broken down as: Euro90 collection, Euro90 sorting, baling Euro40, transport Euro28, wash & dry Euro207, melt process Euro180, additives Euro28 and bagging & warehousing Euro43. Total direct costs are Euro706 and indirect/finance costs are Euro69/tonne. Euro610/tonne can be taken as the market price for recycled material, therefore a shortfall of Euro165/tonne exists between costs and income).

For this study we have taken gate fee estimates from the recent TNO[117]and Prognos[118] reports on plastics recycling in the EU. The ranges of estimates and the figures chosen are shown in Table A7.67.

Table A7.67 Plastic recycling gate fees euro/t

Technique	Range	Selected value
Mechanical recycling	100-500	300
Chemical recycling	50-250	150
Energy recovery	0-150	100

OVERALL GREENHOUSE GAS FLUXES FROM RECYCLING

Overall greenhouse gas fluxes from recycling wastes are summarised in Table A7.68, which recapitulates data presented in the separate sections of this Appendix.

Table A7.68: Emission factors for wastes processed through recycling (kg CO2 eq/t material treated).

Waste management option	**Waste component**	**Short cycle CO_2**	**Fossil CO_2**				**Short cycle C sequestered**	**Sum of fossil C and sequestered C**	**CH_4 emission**	**N_2O emission**	**Total GHG flux**
		(GWP=0)	Process	Energy use	Avoided energy and materials	Transport / mobilisation	**(GWP=-1)**		GWP=21	GWP=310	
	Paper	**0**	**0**	**24**	**-634**	**10**	**0**	**-600**	**0**	**0**	**-600**
	HDPE	**0**	**0**	**24**	**-530**	**15**	**0**	**-491**	**0**	**0**	**-491**
	PET	**0**	**0**	**24**	**-1800**	**15**	**0**	**-1761**	**0**	**0**	**-1761**
	Glass	**0**	**0**	**24**	**-287**	**10**	**0**	**-253**	**0**	**0**	**-253**
	Ferrous metal	**0**	**0**	**24**	**-1521**	**10**	**0**	**-1487**	**0**	**0**	**-1487**
	Aluminium	**0**	**0**	**24**	**-9108**	**10**	**0**	**-9074**	**0**	**0**	**-9074**
	Textiles	**0**	**0**	**24**	**-3203**	**10**	**0**	**-3169**	**0**	**0**	**-3169**

The greenhouse gas flux from recycling materials in 1 tonne of MSW may be estimated by multiplying the above data by the proportion of each material in average MSW, as given in Table 9 on page 25.

WASTE ELECTRICAL AND ELECTRONIC EQUIPMENT (WEEE)

WEEE Recycling Process Description

Approximately 6 million tonnes of waste electrical and electronic equipment (WEEE) is generated each year in Western Europe. At present it represents only 4% of MSW arisings, but the rate of generation is growing 3 times faster than that of MSW [119]. Currently most WEEE is landfilled with a small amount recycled or used for energy recovery. However, forthcoming EC and national directives will increase the amount collected for recycling or recovery.

Table A7.69 shows the arisings of WEEE in the UK in 1998[120]. It should be noted that the figures in this table are believed to be under-estimates. Although IT equipment is a considerable proportion of the WEEE (39%), most of this is from commercial sources and is therefore outside the scope of this study. Smaller household appliances tend to be mainly disposed by householders in the normal household refuse. However, large household equipment can be recycled.

Around half of the WEEE arisings in 1998, mainly large household appliances and IT equipment, were sent to recyclers. However, the amount actually recycled is considerably lower as much of the material sent to recyclers is rejected. For example, 38% of refrigerators and 25% of washing machines sent to scrap fragmentisers were rejected.

Figure A7.46 illustrates the routes to recovery which are taken by large household electrical and electronic appliances. The appliances can be either refurbished for re-sale and re-use, or shredded for material recovery. Re-use is currently limited by the rapid pace of product development. However, promotion of greener product design could reduce this problem.

The UK ICER study found that 47% of the waste generated was ferrous metals, 22% plastics, 6% glass and 4% non-ferrous metals. The main material recovered is ferrous metal. At present it is difficult to recover the glass as most of it arises from TV and computer screens and is contaminated with heavy metals. Mechanical recycling of older plastic components requires strict temperature control during the extrusion process to limit the formation of dioxins and furans from halogenated flame retardents [119]. WEEE can be a valuable source of non-ferrous metals, particularly copper, but most of these are found in computer and telecommunications equipment and electric cables, i.e. mainly in commercial waste rather than MSW. (A recent APME study [119] investigated the feasibility of smelting these waste streams to recover non-ferrous metals, using the plastics content of the waste as an energy source and a reducing agent.)

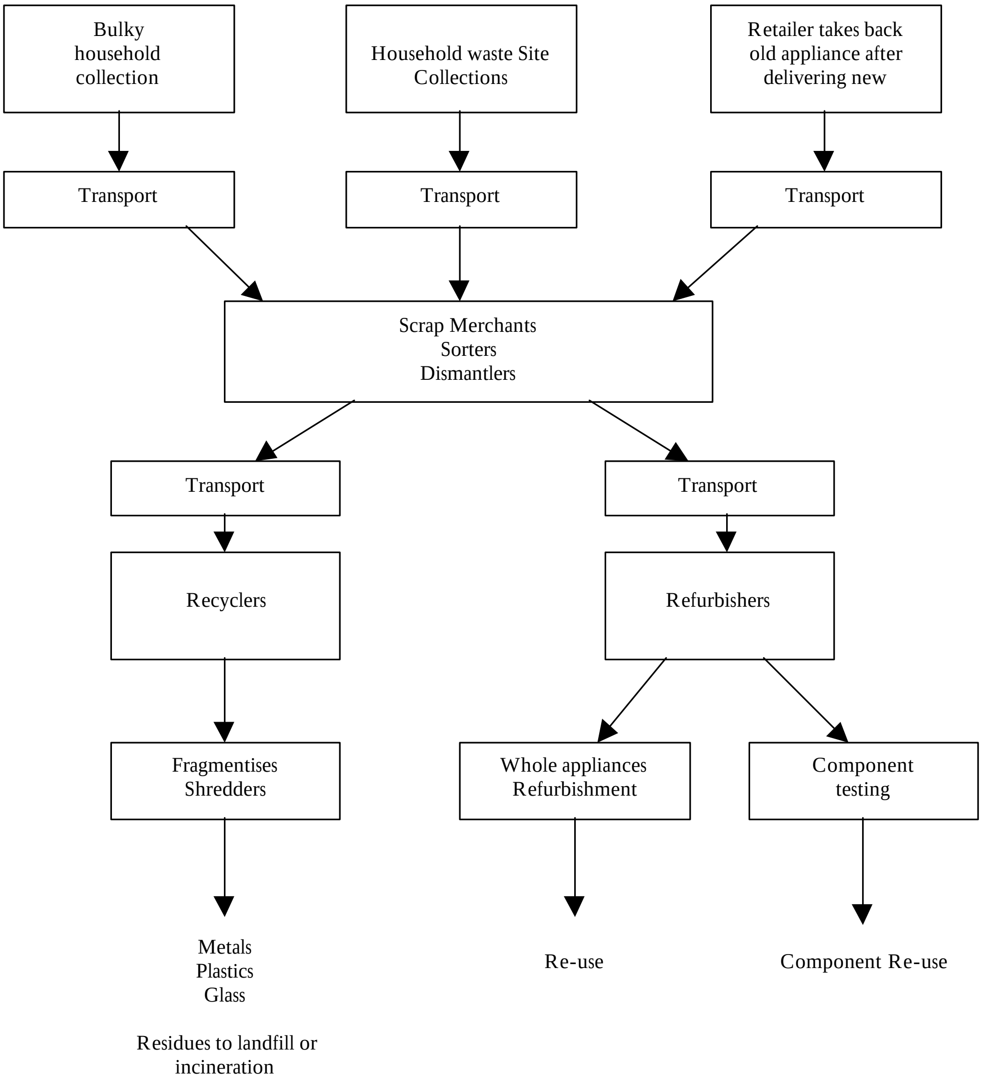
Table A7.69: WEEE arisings in the UK, 1998

Appliance	tonnes	%
Large household appliances	392000	43%
IT equipment	357000	39%
Radio, TV, audio	72000	8%
Small household	30000	3%
Electrical and electronic	28000	3%
Gas discharge lamps	12000	1%
Toys	8000	1%
Monitoring and control	8000	1%
Telecomms	8000	1%
Total	**915000**	
Amount to recyclers	448000	
large household appliances		77%
IT/office equipment		21%

WEEE Recycling Greenhouse Gas Emissions

There are a variety of routes for recycling WEEE. We have taken data from a recent UK study [122] which estimated life cycle emissions from re-use or recycling of various appliances. We have focused on the most common and widespread route at present: recycling of large household white goods equipment via shredding for recovery of ferrous and non-ferrous metal. Other routes may well become more significant in future, as technologies and product design improve.

Figure A7.46 : Recycling routes for WEEE



CFC and HFC Emissions from Refrigerators

Refrigerators and freezers contain greenhouse gases (CFCS and HFCs) as refrigerants and in foam used for insulation. The refrigerant which is in a sealed compressor unit in the fridge is typically CFC 12 in older refrigerators; use of CFC 12 has now been phased out under the Montreal Protocol on Ozone Depleting Substances, and in newer refrigerators has typically been replaced by HFC134a, or in some case by hydrocarbons (propane and isobutane) which are not greenhouse gases. The closed cell foam used as insulation in refrigerators was produced using CFC 11, and as this has been phased out has been replaced by HCFC-141b, or in some cases by pentane or cyclopentane.

CFC refrigerants should (under the Montreal Protocol) be removed and disposed of safely, before the fridge goes on to the next recycling stage (crushing). About 60% of the gas contained in the foam is estimated to be released during disposal121; where refrigerators are being recycled, this would typically be during the shredding phase. The remaining gas in the foam diffuses out during the use of the fridge.

Table A7.70 shows the greenhouse gas emissions which could occur from disposal of an older fridge and newer fridge, in terms of kg CO_2 eq per fridge and per tonne of refrigerators. (Note that newer refrigerators are lighter than older refrigerators as they tend to contain more plastic and less steel parts). With properly controlled disposal, there should be no emissions of the refrigerant,

Table A7.70: Potential CFC and HFC Emissions from Refrigerators

Component	Greenhouse Gas	GWP	kg	Potential Emissions kg CO_2 eq per fridge	kg CO_2 eq per tonne refrigerators
Old fridge					
Refrigerant	CFC12	8100	0.15	1215	32661
Foam	CFC11	3800	0.5	1140	30645
Total weight of fridge			**37.2**		
Total emissions				**2355**	**63306**
New fridge					
Refrigerant	HFC134a	1300	0.06	78	2779
Foam	HCFC 141b	600	0.12	43	1539
Total weight of fridge			**28.07**		
Total emissions				**121**	**4318**

Source: [122,123]. Potential emissions from foam are based on the assumption that 60% of the total burden in the foam is released.

Greenhouse gas emissions from WEEE disposal and recycling

The results shown above for the potential release of CFC and HFC from refrigerators indicate an enormous potential greenhouse gas impact from this component which may make a significant contribution to the overall greenhouse gas impacts of the WEEE stream as a whole.

For the purposes of this analysis, we will compare estimated greenhouse gas fluxes from landfilling or recycling WEEE. The comparison will be made for WEEE containing old refrigerators using the now banned CFCs in their foam and refrigerant, and newer refrigerators using HFC/HCFC. The greenhouse gas impacts of managing WEEE may then be compared with those of managing other components in MSW.

The assumptions made in this analysis are as follows:

- Refrigerators make up 9% of WEEE [122].
- All of the refrigerant is collected before recycling, but all of it escapes after landfilling.
- 60% of the CFC/HFC in the foam escapes during recycling and landfilling, the remainder having escaped during the life time of the product.
- In landfilling, all of the emission is assumed to take place during compaction of the waste. In practice, some will escape after burial and will be collected with landfill gas and destroyed by flaring or combustion in energy recovery systems. However, we assume here that landfills that accept WEEE for disposal are unlikely to have adequate gas control systems in place. All of the CFC/HFC in the refrigerator foam going to landfill will therefore eventually escape into the atmosphere.

- The average composition of WEEE is 47% ferrous metals, 22% plastics, 6% glass and 4% non-ferrous metals [120]. The remaining 26% of the above breakdown is made up of inert material such as concrete, used as a counter-weight in washing machines. Biodegradable materials are assumed to be negligible.
- Recycling involves mechanical shredding of WEEE with other metal scrap, such as end of life vehicles. WEEE in white goods is generally not treated separately at present, although this may be eventually come in if recyclers are required to collect CFC/HFCs released from refrigerator foam. Small items of WEEE (eg telephones, kettles, computers) are not shredded with end of life vehicles. A notional figure of 18 kg CO_2 / tonne of WEEE is used for the shredding step, based on electricity used for shredding of 40 kWh/tonne [124], assuming average EU emission factor (0.45 kg CO_2 / kWh).
- All of the ferrous and non-ferrous metals are recovered after shredding for recycling. The remaining material is either landfilled directly or plastic is separated for incineration with CHP energy recovery.
- Recycling of the ferrous metals results in savings of 1810 kg CO_2 equivalent /tonne of metal, compared with manufacture from primary source, as described in the section on metal recycling.
- Aluminium and copper are the main components of the non-ferrous metal in WEEE. We assume an equal amount of each.
- Recycling of aluminium results in saving of 9837 kg CO_2 equivalent /tonne of metal, compare with manufacture from primary sources, as described in the section on metal recycling.
- Recycling of copper saves emissions of 15,400 kg CO_2 equivalent / tonne of metal compared with manufacture from virgin sources. This figure was calculated from the energy consumption needed to produce copper from medium grade ore (containing 0.3-1% Cu), quoted as 91 – 184 GJ/tonne [125]. Reprocessing of copper scrap use between 4 and 40% of the energy needed to produce the metal from primary sources [126], depending on the quality of the scrap. We assume that the copper in WEEE is available in a relatively clean form and so requires 10% of the energy for reprocessing compared with primary production. Taking a mid range value of 137 GJ/tonne for production from primary ore, assuming 90% of this can be saved by recycling copper from WEEE and using a CO_2 emission factor of 0.45 kg CO_2 /kWh gives a net saving due to copper recycling of 15,400 kg CO_2 /tonne of copper.
- Plastics recovered from the shredded WEEE are assumed to be either landfilled (in which case they have no greenhouse gas impact) or incinerated in a mass-burn incinerator with CHP energy recovery. The net greenhouse gas impacts due to incinerator emissions and displaced energy (at EU average plant mix) equate to 308 kg CO_2 equivalent/tonne of plastic (see Appendix 3).
- Greenhouse gas fluxes associated with transport and landfill operations are negligible in comparison with fluxes from other sources and have been omitted from this analysis of WEEE.

On the basis of these assumptions, we can now estimate the total greenhouse gas impacts from recycling or landfilling WEEE containing ‘old’ or ‘new’ refrigerators. The results are shown in Table A7.71.

Table A7.71: Estimated greenhouse gas emissions from recycling or landfilling WEEE with refrigerators containing CFCs ('Old') or HFCs ('New').

Process / source	Old – recycle	New – recycle	Old - Landfill	New - Landfill
Shredding	18	18	n/a	n/a
Refrigerants / foam	2,758	139	5,698	389
Ferrous metal recycling	-760	-760	0	0
Copper recycling	-308	-308	0	0
Aluminium recycling	-197	-197	0	0
Plastic to incinerator with CHP	66	66	0	0
Total without plastic incineration	1,511	-1,109	5,698	389
Total with plastic incineration	**1,577**	**-1,042**	**5,698**	**389**

The results show that on a per tonne basis, landfilling 'old' WEEE is potentially the biggest source of greenhouse gas emissions from the wastes considered so far. The implementation of the WEEE Directive will reduce landfilling of this waste substantially in the future, and the phase out of CFCs will further reduce emissions. However, given the service lifetime of refrigerators of about 10 to 15 years, a significant pool of CFCs is still waiting to come through into the waste stream. Diversion of WEEE to recycling, with removal and destruction of CFCs prior to recycling, will do much to reduce emissions from this source. Newer technologies are also being developed, such as shredding or crushing foam in sealed equipment or under oil, so that the CFCs may be recovered and destroyed. However, these are not yet widely deployed and will add substantially to the costs of refrigerator recycling, since these products will require special treatment in dedicated shredders at the reprocessors, rather than being processed with other white goods and end of life vehicles as at present.

WEEE Recycling Costs

The economic and environmental impacts of recovery of WEEE were investigated in June 1997[127]. This report examined a number of pilot collection trials being carried out in the Member States and summarised that collection costs were in the range Euro180-2445 per tonne. The analysis showed that collection systems based on community or local authority collection systems were the cheapest (usually in the range Euro450-860 per tonne). Retailer return schemes were more expensive (>Euro1000 per tonne). The report also found that the costs of reprocessing WEEE varied widely depending on type of item and its complexity. Costs ranged from about Euro170 per tonne for mainframe computers to Euro600 per tonne for TV sets. The study indicated that integrated reprocessing systems such as that operated by Mann UK could reduce costs of reprocessing to about Euro100 per tonne.

The study also conducted a 'bottom up' cost analysis based on a typical set up for reprocessing WEEE collected from a typical catchment area (Table A7.72).

Table A7.72 - Summary of Results from the 'Bottom-up' Analysis

Cost Type	Cost per tonne (Euro)	
	Current	Future
Transport costs	103	61
Other collection costs	52	30
Reprocessing costs	122 – 144	109 – 125
Residue disposal costs	70	84
TOTAL	347 – 369	284 - 300

This analysis of costs of WEEE recovery and recycling was based on the results from several pilot trials throughout the Member States. Costs of recovery and recycling of non-WEEE items have reduced significantly when the scale of operations have been increased allowing economies of scale, and methods have been optimised though experiences gained. No doubt this will also be the case for WEEE recovery and recycling – as experience is gained, improvements in efficiency can be expected to reduce costs per tonne.

The latest draft of the proposed WEEE directive was issued by on 10th May 2000. The Explanatory Memorandum attached to it contains an economic assessment which quotes ranges of costs for collection and reprocessing, based on a number of studies, including the 1997 study quoted above. The quoted costs, translated to cost per tonne figures, are shown in Table A7.73:

Table A7.73 Estimated Costs for WEEE Collection and Reprocessing

Activity	Cost per tonne (Euro)
Collection costs	200-400
Recycling costs	
Large household equipment	10-80
Refrigerators	200-300
Equipment with monitors	100-800
Small household equipment	200-500

Appendix 8. References

1 AEA Technology (1998) Options to reduce methane emissions. Report to DG XI of the European Commission.

2 IPCC (1996) Climate change 1995. *The Science of Climate Change*. Summary for policymakers and technical summary working group report.

3 AEA Technology (1998) Options to reduce nitrous oxide emissions. Report to DG XI of the European Commission.

4 United Nations Framework Convention on Climate Change (1995) – the Kyoto Protocol. http://www.unfccc.int/resource/process/components/response/respkp.html

5 European Commission (2001) Biological treatment of biowaste – Working document 2nd draft. http://europa.eu.int/comm/environemt/waste/facts_en.htm

6 Intergovernmental Panel on Climate Change, 2000, ‘Land Use, Land Use Change and Forestry, Summary for Policy Makers’, A Special Report of the Intergovernmental Panel on Climate Change.

7 Revised 1996 IPCC Guidelines for national Greenhouse Gas Inventories: Reference Manual.

8 European Commission, 1995, ‘ExternE Externalities of Energy, Vol 6, Wind and Hydro’, table 6.5, EUR 16525 EN, ISSN 1018-5593.

9 European Commission, 1999, ‘ExternE Externalities of Energy, Vol 10, National Implementation’, tables 6.12 and 10.10, EUR 18528 EN, ISBN 92-828-3723-8.

10 OECD 1999, ‘OECD Environmental Data 1999’, Chapter 7, Waste.

11 APME (Association of Plastics Manufacturers in Europe) (1999) Information system on plastic consumption and waste management in Western Europe. European overview – 1997 data. Report by Taylor Nelson Sofres Consulting.

12 CEPI (Confederation of European Paper Industries) Newsletter 13, August 1999.

13 APEAL press release ‘Recycling of steel packaging in Europe in 1999’, www.apeal.org.

14 ENDS Daily 15 September 2000, ‘European steel can recycling rebounds in 1999’.

15 European Aluminium Association, ‘Aluminium beverage can consumption and recycling www.aluminium.org

16 Jim Poll, AEA Technology, personal communication 26/3/01.

www.apme.org.

24 ENDS Daily 6 June 2000, ‘Swedish waste recycling up in 1999’. www.ends.co.uk.

25 ENDS Daily 3 May 2000, ‘Sharp fall in EU plastics waste landfilling’. www.ends.co.uk.

26 ENDS Daily 13 April 2000, ‘PVC waste treatment options surveyed’. www.ends.co.uk.
27 S Latham and G Mudge, 1997, ‘Life Cycle Research Programme for Waste Management: Inventory development for waste management operations - waste transport and other vehicle use (final report)’. TRL Report.

28 Favoino E et al, 2000. ‘Sorting Food Waste in Italy: optimising costs and benefits’, Warmer Bulletin 70, pp 4-7, January 2000.

29 DHV Environment and Infrastructure, 1997; ‘Composting in the European Union, Final Report’, DHV Report AT-973090, June 1997.

30 AEA Technology 1999, ‘Cost-effectiveness of Options for reducing UK methane emissions’, AEAT-4692.

31 Information provided by national experts to European Commission DG TREN, March 2000, file .

32 AEA Technology 2000, ‘Economic evaluation of PVC waste management’, report for DG Environment of the European Commission.

33 Colin Williamson, 1992, Plastics & Rubber Weekly 14/3/92.

34 Patel M et al, 2000, 'Recycling of plastics in Germany', Resources Conservation & Recycling 29 pp65-90.

35Andrew Simmons, RECOUP, Private communication.

36 Materials Recycling Week, 5 June 1998, p28.

37 Christiansen, TH and Kjedsen, P (1989). Basic biochemical processes in landfills. In: *Sanitary landfilling: Process, technology and environmentalimpact*. (Eds TH Christiansen, R Cossu and R Stegmann), Academic Press, London, UK pp 29-44.

38 AEA Technology (1999). Methane emissions from UK landfills. A report for the UK Department of the Environment, Transport and the Regions, by KA Brown et al.

39 Bognor, JE, Matthews, E, Katzebstein, A, Blake, D and Carolan, M (2000). Greenhouse gas emissions from landfills: What we know and what we don't know. In:*ISAW World Congress Proceedings. 2000 and beyond – which choices for waste management*. Paris France 3-7 July 2000. pp 421-430.

40 IPPC (1996). *Climate change 1995 – The Science of Climate Change: Contribution of working group 1 to the second assessment report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, UK.

41 Philips Geographical Digest 1994-5, projections for 2000.

42 Personal communication from David Whiteleg, Shanks & McEwan Waste Management, to Keith Brown, January 2001.

43 Intergovernmental Panel on Climate Change (IPCC) (1996). IPCC Guidelines for National Greenhouse Gas Inventories: Reference Manual – Chapter 6: Waste.

44 Environment Agency. (1994). National Household Waste Analysis Project Phase 2 Volume 1: Report on composition and weight data. CWM 082/94. Volume 2: Report on further composition and weight data. CWM 086/94. Volume 3: Chemical analysis data. CWM 087/94.

45 UK Department of the Environment Transport and the Regions, 1999, 'A Way with

46 Stegmann, R, Ehrig, H-J, & Rettenberger, G (2000). Landfill gas formation, quality and prediction. In: Asian Landfill Symposium 2000 – Solid Waste Landfill.

47 Rathje WL et al, 1989, 'Inside Landfills. A preliminary report of the Garbage Project's 1987-88 excavations at five landfills', presented at Municipal Solid Waste Technology conference, San Diego, January 1989.

52 Reimann, DO (2000) Waste management and prospects for energy from waste in Germany. Presentation to the International Energy Agency Task 23 (Energy from Thermal Conversion of MSW and RDF) meeting in Karlsruhe, Germany, 1-3 November 2000.

53 Schöen LAA, Beekes ML, van Tubergen J and Korevaar CH, 'Mechanical separation of mixed plastics from household waste and energy recovery in a pulverised coal fired power station', APME technical paper. www.apme.org.

54 Corey RE (ed) (1969) Principles and practices of incineration. Wiley Interscience, NY.

55 Paar, S, Brummack, J and Gemende, B. (1999). Advantages of dome aeration in mechanical-bilogical waste treatment. In: *Proceedings Sardinia 99. Seventh International Waste Mangement and Landfill Symposium. S Margherita di Pula, Caligari, Italy, 4-8 October 1999.* pp 329-334.

56 Favoino, E. (2001) personal communication.

57 TA Siedlungsabfall, Bonn, 1993.

58 ENDS Daily 31/1/01, 'German mech-bio waste law approved'.

59 Soyez, K, Koller, M and Thaen, D. (1999) Mechanical –biological pretreatment of residual waste: Results of the German Federal Research Programme. In: *Proceedings Sardinia 99. Seventh International Waste Mangement and Landfill Symposium. S Margherita di Pula, Caligari, Italy, 4-8 October 1999.* pp 379-386.

60 Raninger, B, Pilz, G and Gheser, D. (1999). Optimisation of mechanical-biological treatment of waste to achieve Austrian landfill requirements. In: *Proceedings Sardinia 99. Seventh International Waste Mangement and Landfill Symposium. S Margherita di Pula, Caligari, Italy, 4-8 October 1999.* pp 387-394.

61 Hoering, K, Kruempelbeck, I and Ehrig, H-J. (1999) Long-term emission behaviour of mechanical-biological pre-treated municipal solid waste. In: *Proceedings Sardinia 99. Seventh*

International Waste Mangement and Landfill Symposium. S Margherita di Pula, Caligari, Italy, 4-8 October 1999. pp 409-418.

62 Rettenberger, G. (1999) New landfill technology for mechanically biologically pretreated waste. In: *Proceedings Sardinia 99. Seventh International Waste Mangement and Landfill Symposium. S Margherita di Pula, Caligari, Italy, 4-8 October 1999.* pp 527-532.

63 Federal Environment Agency – Austria, 1998, 'Abluftemissionen aus der mechanisch-biologischen abfallbehandlung in Osterreich', Umweltbundesamt monographien/Bd. 104, ISBN 3-85457-445-2.

64 Hackl, A and Mauschitz, G, 2000, 'Greenhouse Gas Mitigation by Proper Waste Management', study for the Austrian Federal Ministry for Environment, Youth and Family Affairs.

65 Wannholt L, 1998, 'Biological treatment of domestic waste in closed plants in Europe - Plant visit reports', RVF report 98:8 for the Swedish Association of Solid Waste Management.

66 Ecofys, NTUA, AEA Technology 2000, 'Options to reduce methane from landfills' chapter from Economic Evaluation of Sectoral Emission Reduction Objectives for Climate Change, report in preparation for DG Environment.

67 US EPA 1998, 'Greenhouse Gas Emissions from Management of Selected Materials in Municipal Solid Waste, EPA530-R-98-013.

68 IPCC (1996) *Climate change 1995. Impacts, adaptations and mitigation of climate change: Scientific and technical analysis.* Intergovernmental Panel on Climate Change, Cambridge University Press, New York.

69 ECAF (not dated) *Conservation Agriculture in Europe: Environmental, Economic and EU Policy Perspectives.* European Conservation Agriculture Federation, Brussels.

70 Schlesinger, WH (1995) An overview of the carbon cycle. *In: Soils and Global Change.* Eds R Lal et al. CRC Lewis Publishers, Boca Raton, Florida, pp9-25.

71 Felipo, MT (1996) Compost as a source of organic matter in Mediterranean soils. *In: The Science of Composting.* Eds M de Bartoldi et al pub Blackie Academic & Professional.

72 Jenkinson, DS (1988) Soil organic matter and its dynamics. *In: Soil conditions and plant growth.* Ed. A Wild, 11th edition.

73 Hodges, RD (1990) Soil organic matter – its central position in organic farming. *In: Advances in soil organic matter research: The impact on agriculture and the environment.* Eds RD Wilson et al. Royal Society of Chemistry. pp355-363.

74 Jenkinson, DS (1988) Soil organic matter and its dynamics. *In: Soil conditions and plant growth.* Ed. A Wild, 11th edition.

75 Crawford, RL & Crawford, DL (1984) *Enzyme Microbial Technol*. **6** 434.

76 Martin, JP & Haider, K (1972) *Soil Biol & Biochem*. **4** 245.

77 Paustian, K, Parton, WJ & Persson, J (1992) Modelling soil organic matter in organic amended and nitrogen-fertilised long-term plots. *Soil Sci Soc Amer J*. **56** 476-488.

78 Various authors in *Sustainable management of soil organic matter*. (eds RM Rees *et al*). Based on papers offered at a meeting held in Edinburgh September 1999. Pub CABI Publishing. ISBN 0 85199 465 2. (2001).

79 Parton, WJ, Stewart, JW & Cole, CV (1988) Dynamics of C, N, P and S in grassland soils – a model. *Biogeochemistry* **5** 109-131.

80 Carter MR, 'Organic matter and sustainability', in 'Sustainable management of soil organic matter', ed. Rees RM et al, based on papers offered at a meeting in Edinburgh 1999. CABI publishing. ISBN 0 85199 465 2.

81 Smith, P, Powlson, DS, Glendining, MJ, Smith JU (1997) Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using long-term experiments. *Global Change Biology*, **3**, 67-79.

82 Liners, M, Juste, C, Tanzin, J and Gomez, A (1986) Effect on long-term sludge disposal on soil organic matter characteristics. In: *Processing and use of organic sludge and liquid agricultural wastes*. (PL Hermite Ed.) Reidel, Dordrecht, 290-303.

83 Hauke, H & Stoeppler-Zimmer, R (1996) Development of compost products. *In: The Science of Composting*. Eds M de Bartoldi et al pub Blackie Academic & Professional.

84 Ward, S (2000) Peat as a biofuel. Bord na Mona web site www.bnm.ie/exploring/peat_as_a_bio_fuel2.html.

85 Ward, S (2001) pers comm to KA Brown.

86 Peat Producers' Association

87 Johnston AE et al 1997, 'Peat – A valuable resource', in Hayes MHB and Wilson WS, Humic substances in soils, peats and waters: Health and Environmental Aspects', Royal Society of Chemistry, UK.

88 Crill, P, Hargreaves, K & Korhola A (2000) The role of peat in Finnish greenhouse gas balances. Report commissioned by the Ministry of Trade and Industry, Finland, ISBN 951-739-542-6.

89 Friends of the Earth, 'Last chance to see…? Lowland raised bog SSSIs threatened by commercial peat extraction', www.foe.co.uk/wildplaces.

90 Sikora, LJ (1996) Effect of compost on crop growth. *In: The Science of Composting*. Eds M de Bartoldi et al pub Blackie Academic & Professional.

91 Baldoni, G, Cortellini,L & Dalre, L (1996) The influence of compost & sewage sludge on agricultural crops. *In: The Science of Composting*. Eds M de Bartoldi et al pub Blackie Academic & Professional.

92 Rodrigues, MS, Lopez-Real, JM & Lee, HC (1996) Use of composted societal organic wastes for sustainable crop production. *In: The Science of Composting*. Eds M de Bartoldi et al pub Blackie Academic & Professional.

93 Allievi, L et al (1993) Plant quality and soil residual fertility six years after a compost treatment. *Bioresource Technology*, 43, 85-89.

94 Maynard, AA (1993) Nitrate leaching from compost-amended soils. *Compost Science and Utilisation*, 1(2), 65-72.

95 Cortellini, L, Toder, G, Baldoni, G,& Nassisi, A (1996) Effects of the content of organic matter, nitrogen, phosphorus and heavy metals in soil and plants after application of compost and sewage sludge. *In: The Science of Composting*. Eds M de Bartoldi et al pub Blackie Academic & Professional.

96 Parkinson, R, Fuller, M, Jury,S &Groenhof, A (1996) An evaluation of of soil nutrient status following application of co-composted MSW and seawge sludge and greenwaste to maize. *In: The Science of Composting*. Eds M de Bartoldi et al pub Blackie Academic & Professional.

97 Pinamonti, F, & Zorzi, G (1996) Experience of compost use in agriculture and land reclamation projects. *In: The Science of Composting*. Eds M de Bartoldi et al pub Blackie Academic & Professional.

98 Wallace PA (1996) Field trials of compost for agriculture. Report CWM 158/96.

99 DoE (1996). Markets and quality requirements for composts and digestates from the organic fraction of household waste. *A report to the UK Department of the Environment and Environment Agency*. Prepared by Warren Spring Laboratory, Shanks & McEwan (Energy Services) Lt and David Border Composting Consultancy. CWM 147/96.

100 AEA Technology (1998) Options to reduce nitrous oxide emissions. Report to the European Commission DG XI. http://europa.eu.int/comm/environment/enveco/climate_change/nitrous_oxide_emissions.pdf

101 Kongshaug G, Hydro Agri Europe, Norway, 'Energy consumption and greenhouse gas emissions in fertilizer production', EFMA seminar on EU legislation and the legislation process in the EU relative to fertilizer, Prague, October 1998.

102 Biermann, S, Rathke, G-W, Huelsbergen, K-J and Diepenbrock, W (1999) Energy recovery by crops in dependence on the input of mineral fertiliser. Final report to the European Fertiliser Manufacturers"Association.

103 Mosier, AR (1993) Nitrous oxide emissions from agricultural soils. *In: Proceedings of methane and nitrous oxide - methods of national emissions inventories and options for control* RIVM, Netherlands.

104 DHV Environment and Infrastructure 1997, 'Composting in the European Union', report for DGXI of the European Commission, reference AT973090.

105 Novem 1992, 'Conversion techniques for VGF biowaste', report 9317.

106 AEA Technology 2000, 'Implications of the EC Landfill Directive and the Draft Waste Strategy on UK GHG: Preliminary Study', report for the DETR. AEAT/EPSC-0090.

107 Coopers & Lybrand, 1997, 'Cost-benefit Analysis of the Different MSW Management Systems: Objectives & Instruments for the Year 2000, Report to EC DG Environment.

108 Recycling Times, 8 September 1992.

109 European Environment Agency 1999, 'Environment in the EU at the turn of the century', chapter 3.7.

110 BNMA 1995, 'Recycle or Incinerate? The future for used newspapers: an independent

111 Swiss Agency for the Environment, Forests and Landscape (SAEFL), Berne, 1998, 'Life Cycle Inventories for Packagings', Environmental Series no. 250.

112 Materials Recycling Week

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

114 Chem Systems, 1997. 'Life Cycle Inventory Development for Recycling', Environment Agency Project Record, Environment Agency , UK

115Andrew Simmons, RECOUP, Private communication.

116 Colin Williamson, 1992, Plastics & Rubber Weekly 14/3/92.

117 TNO, 2000, 'Chemical recycling of plastics wastes (PVC and other resins).

118 Prognos 2000, Mechnical recycling of PVC wastes, Study for DG XI of the European Commission (B4-3040/98/000821/MAR/E3)

119 Mark FE and Lehner T, 'Plastics recovery from Waste Electrical and Electronic Equipment in Non-ferrous metal processes', APME report, www.apme.org/literature.

120 ICER, 2000. 'UK Status Report on Waste from Electrical and Electronic Equipment'.

121 March, 1999. 'UK Emissions of HFCs, PFCs and SF_6 and Potential Emission Reduction Options', DETR, UK.

122 Ecobalance, 1999. 'Life Cycle Assessment and Life Cycle Financial Analysis of the Proposal for a Directive on Waste from Electrical and Electronic Equipment'.

123 Poll AJ, 1993. 'The Recycling and Disposal of Domestic Electrical Appliances', Warren Spring Laboratory Report LR980.

124 Personal communication from David Hulse, British Metals Federation, to Keith Brown, March 2001.

125 Chapman & Roberts (1983) Metal Resources and Energy. Pub Butterworths, Boston, Mass. p138.

126 Kirk Othmer Encyclopedia of Chemical Technology

127 AEA Technology, 1997, 'Recovery of WEEE: Economic and Environmental Impacts', AEA Technology Report for European Commission DGXI.

14 KH-40-01-432-EN-N

Environment themes

General

Water

Land

Air

Industry

Waste

Nature

Urban

Funding

Law

Economics

Assessment

Nuclear issues

Risks

Education

See our publications catalogue at:
http://europa.eu.int/comm/environment/pubs/home.htm

OFFICE FOR OFFICIAL PUBLICATIONS
OF THE EUROPEAN COMMUNITIES

L-2985 Luxembourg

ISBN 92-894-1733-1

9 789289 417334