ECONOMIC ANALYSIS OF OPTIONS FOR MANAGING BIODEGRADABLE MUNICIPAL WASTE

Final Report to the European Commission

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Preface

We are grateful to the many individuals who have assisted in the preparation of this report. In particular, we would like to thank Pierre Strosser, Sonia Fumagalli and Luca Marmo of the European Commission for their helpful comments throughout the course of the report's preparation.

ABBREVIATIONS

AD	Anaerobic Digestion
BMW	Biodegradable Municipal Waste
CA site	Civic Amenity Site
CAP	Common Agricultural Policy
CCGT	Combined Cycle Gas Turbine
CHP	Combined Heat and Power
FEAD	Féderation Européenne des Activités du Dechet et de l'Environment
GDP	Gross Domestic Product
GMOs	Genetically Manipulated Organisms
IPPC	Integrated Pollution Prevention and Control
MBT	Mechanical Biological Treatment
MHV	Medium Heating Value
MSW	Municipal Solid Waste
NFFO	Non-Fossil Fuel Obligation
OECD	Organisation for Economic Co-operation and Development
PCBs	Poly-chlorinated Biphenyls
PFI	Private Finance Initiative
PRNs	Packaging Recovery Notes
RCVs	Refuse Collection Vehicles
RDF	Refuse-Derived Fuel
RVF	Swedish Association of Waste Management

SO	Standards Only (scenario)
SP	Standards Plus (scenario)
TEQs	Toxic Equivalents
VOCs	Volatile Organic Compounds

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Final Report to the European Commission

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1.0 INTRODUCTION AND SCOPE

ECOTEC Research and Consulting Limited (ECOTEC), in association with Eunomia Research & Consulting, HDRA Consultants Ltd (UK), Zentrum für Rationelle Energieanwendung und Umwelt GmbH (ZREU) (Centre for Rational Use of Energy and Environment Ltd.) (Germany), Scuola Agraria del Parco di Monza (Italy), and LDK Consultants (Greece), has been asked by the European Commission to carry out an Economic Analysis of Options for Managing Biodegradable Municipal Waste. This takes place at a time when many countries, especially those that are heavily dependent upon landfill, are considering options of this nature in the context of the Article 5 targets in the Council Directive on the Landfill of Waste (the Landfill Directive).[1]

1.1 Aims and Objectives

The main objective of the study is:

To conduct an economic evaluation, that considers both private and social welfare costs and benefits, of existing options for managing the biodegradable fraction of municipal solid waste.

Although all management options (anaerobic digestion, composting, landfilling, incineration, etc.) are considered in the study, the main emphasis is on the separate collection and recycling of the biodegradable fraction of MSW. The study focuses on the Member States of the European Union and on the first wave of Accession countries, i.e. the Czech Republic, Poland, Hungary, Estonia, Slovenia and Cyprus.

Specific tasks are:

- To identify the main private and public stakeholders that are involved in the management and recycling of organic municipal solid waste (MSW), along with their main technical, social, financial / economic and legal constraints

- For the different management options (anaerobic digestion, composting, land-filling, incineration etc.) that are currently practised, and in particular, for the separate collection and recycling options that will be identified, to specify:

    a) the technical problems and primary constraints related to the implementation of the option;

    b) the quantity, the type and the quality of waste that may potentially be targeted by the option;

[1] OJ L 182/1, 26.4.1999.

c) the characteristics and quality of the end product, and its potential markets (types of use, market size, marketing constraints);

d) the quality and characteristics of any final residues generated by the treatment option, and any issues associated with their management and treatment;

e) the financial and economic costs and benefits of each option;

f) for the different quantitative figures provided, to give a range according to the most distinct observed situations (urban versus rural areas, different feedstock, different physical environment or states, etc)

- To compare the costs of alternatives to the end-products (bio-methane versus other sources of energy, nitrate/phosphate to plants from different types of composts versus manure or fertilisers, etc)

- Based on the results of the above mentioned tasks, to undertake the economic (cost-benefit) analysis of possible changes in EU legislation for managing biodegradable municipal waste in the different Member States and the entire European Union

- To perform appropriate sensitivity analyses on the main assumptions made for estimating costs and benefits / elements and for undertaking the cost-benefit analysis of the implementation of the proposed Directive (implementation scenario on the Directive, technological change, Common Agricultural Policy (CAP) reform, etc.)

- To identify and qualify any significant potential internal market and trade issues that may arise as a result of different National legislation on composting, and to assess the impact of harmonised legislation on these issues.

1.2 Scope of the Analysis

1.2.1 Definition of Municipal Waste

It was intended that the study should use the definition of 'municipal waste' as it is used in specific Member States. Definitions of 'municipal waste' differ greatly between countries, as is made clear in a report by the University of Louvain-la-Neuve Business School (1998), who make the point that: '*While almost all municipalities have responsibility for the management of household waste, the definition and responsibility of non-household municipal solid waste, or industrial waste, or construction waste vary greatly from system to system [between countries].*' In addition, the European Topic Centre on Waste has noted the discrepancies in Member State definitions, as well as the fact that the terms 'household waste' and municipal waste' are often used as though the two were interchangeable, even though they are not the same thing (Christiansen and Munck-Kampmann 2000).

The problems associated with the different definitions are magnified when one tries to understand the composition of municipal waste. This is because it is not always obvious what it is whose composition is being measured. For example, where composition relates to waste collected at the doorstep of households, this may be a very poor approximation to the composition of municipal waste where large amounts of commercial waste are collected, or where households make extensive use of civic amenity sites or containerparks.

1.2.2 Policy Changes Being Considered

No policy measure has yet been agreed by the European Commission concerning biodegradable municipal wastes other than the Landfill Directive itself. Two policy variants are considered in this report. These can be characterised as:

- Standards-only (SO) a policy which establishes only technical standards for materials collection and composting processes (in terms of, for example, heavy metals content); and
- Standards-plus (SP) a policy which not only establishes standards, but which also puts in place requirements for the separate collection of, and / or home composting of, biodegradable municipal waste.

For the sake of argument, and for purposes of clarity, these are referred to as the 'standards-only' (SO) and 'standards-plus' (SP) scenarios. The focus is very much on the SO scenario for two reasons. In the first instance, the effect of a standards only policy is much less straightforward to predict. Secondly, the SP policy has more certain outcomes and lends itself more readily to quantification of impacts, though these are by no means straightforward to estimate.

1.2.3 Biodegradable Wastes Considered in 'Standards Plus' Scenario in the Study

For purposes of clarification, in the SP scenario, only 'biowastes' are being considered. Other biodegradable wastes such as paper, textiles and nappies are not part of the study's key focus. Biowastes as defined here include:

- Kitchen wastes from households
- Yard wastes from gardens; and
- Where included in the definition of 'municipal waste', kitchen wastes from restaurants, green waste from parks, and wastes similar to kitchen wastes / parks from commerce and industry.

The choice reflects the principal intent of any possible new policy instrument that may be introduced, and the focus of the current study in terms of treatments.

It is appreciated that significant quantities of paper can be composted, but composting of paper is not a focus of this study. In addition, it is appreciated that biological treatment methods can deal with more wastes than those which are the primary focus here. However, the study was asked to focus on those fractions which are most commonly targeted for biological treatments post-separation.

Compositional data should help to identify how much biodegradable waste other than "biowaste" (as defined above), especially paper, is present in MSW. This is important since in this study, the impact of the proposed policy change has to be measured against a baseline. The baseline is now, effectively, the situation as it will look under the Landfill Directive. The key part of the Directive from the point of view of this study is Article 5 (2), which is shown in Box 1.

Other aspects of the Directive will have an impact on the options used to treat municipal waste. For example, where Member States do not have landfills dedicated to the treatment of hazardous waste only, landfilling of hazardous wastes will only be able to continue if such landfills are established. Hence, it may become more difficult to find outlets for fly ash from incinerators and their disposal may become more costly, depending upon the way in which the Landfill Directive is transposed into Member State legislation.

BOX 1: ARTICLE 5 (2) OF THE COUNCIL DIRECTIVE ON THE LANDFILLING OF WASTE

2. This strategy shall ensure that:

not later than five years after the date laid down in Article 18(1), biodegradable municipal waste going to landfills must be reduced to 75% of the total amount (by weight) of biodegradable municipal waste produced in 1995 or the latest year before 1995 for which standardised Eurostat data is available.

not later than eight years after the date laid down in Article 18(1), biodegradable municipal waste going to landfills must be reduced to 50% of the total amount (by weight) of biodegradable municipal waste produced in 1995 or the latest year before 1995 for which standardised Eurostat data is available.

not later than 15 years after the date laid down in Article 18(1), biodegradable municipal waste going to landfills must be reduced to 35% of the total amount (by weight) of biodegradable municipal waste produced in 1995 or the latest year before 1995 for which standardised Eurostat data is available.

Two years before the date referred to in paragraph © the Council shall re-examine the above target, on the basis of a report from the Commission on the practical experience gained in Member States in the pursuance of the targets laid down in paragraphs (a) and (b) accompanied, if appropriate, by a proposal with a view to confirming or amending this target in order to ensure a high level of environmental protection.

Member States which, in 1995 or the latest year before 1995 for which standardised Eurostat data is available, put more than 80% of their collected municipal waste to landfill may postpone the attainment of one or more targets et out in paragraphs (a), (b) or (c) by a period of not exceeding four years. Member States intending to make use of this provision shall inform the Commission of their decision in advance. The Commission shall inform other Members States of these decisions.

The implementation of the provisions set out in the preceding subparagraph may in no circumstances lead to the attainment of the target set out in paragraph (c) at a date later than four years after the date set out in paragraph (c).

1.2.4 Outline of Approach to the Study

The way in which the study has been carried out is represented schematically in Figure 1. Essentially, there are 6 steps in the process:

- **Step 1:** This involves analysing the external costs and benefits of the different options. The bulk of this work is carried out in Section 4 and in the related Appendices. The shortcomings and omissions in this analysis are identified;

- **Step 2:** This assesses the private costs of the different treatment options. A gate fee approach has been adopted, which is less than ideal for this type of analysis. This work is based on country investigations and is carried out in Section 5;

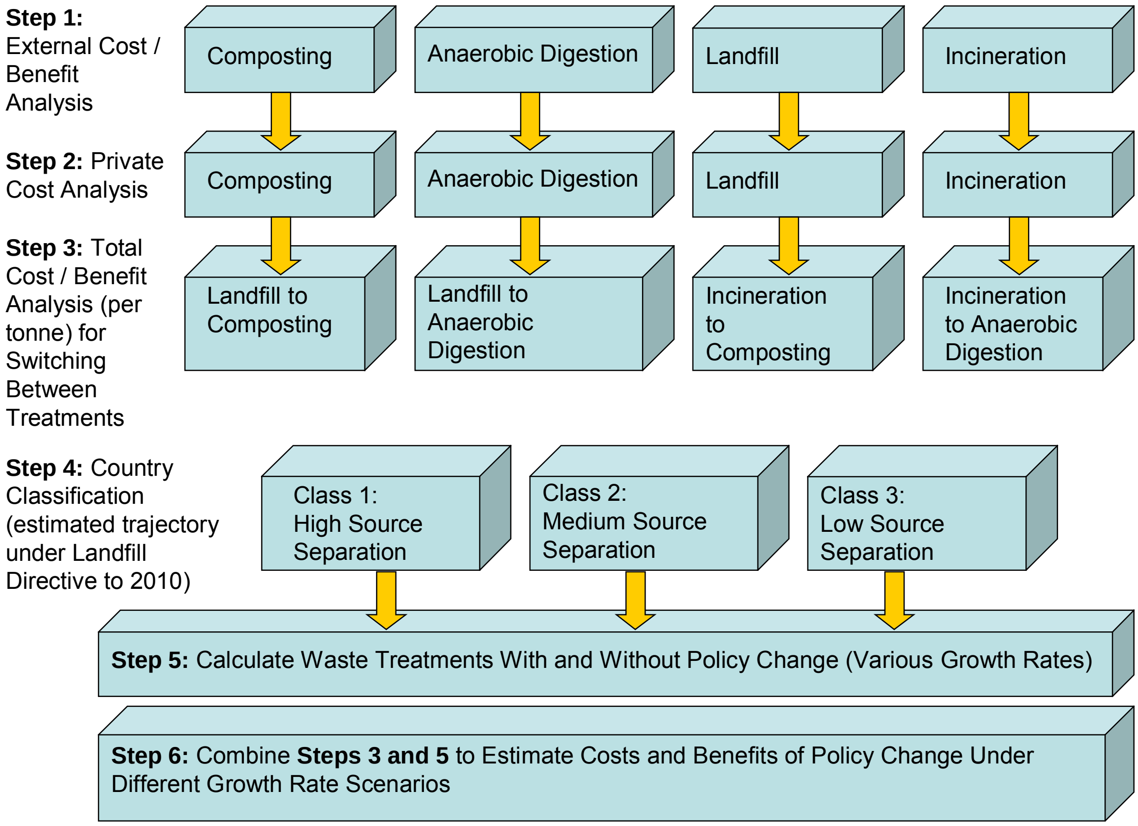
- **Step 3:** Based on Steps 1 and 2, the private and external costs and benefits of changing fro landfill or incineration, to either composting or anaerobic digestion are estimated. This analysis is carried out for each individual country and the results are outlined in Section 6;

- **Step 4:** Based on investigations of the current situation (Section 3) and Member State (and Accession state) plans, countries are assigned a country classification based on how far they are expected to have achieved source separation of wastes for composting or digestion by the year 2010;

- **Step 5:** These country classifications and associated scenarios (Section 7) are used to estimate the amount of waste going to each of the key treatment routes in 2010 under a scenario in which only the Landfill Directive in place. This is done for different rates of growth in municipal waste. The changes in the amount of waste going to each of the treatment routes implied by a policy recommending source separation are then estimated, the extent of the change being related to the country classification;

- **Step 6:** The final step involves bringing the results of Steps 3 and 5 together (Section 7). The estimated movement of waste away from landfill and incineration and towards composting and anaerobic digestion is combined with the unit values of costs and benefits associated with switching between treatments. These two pieces of information enable one to derive an estimate of total costs and benefits of the policy change proposed.

Figure 1: Schematic Outline of Approach



1.3 Outline of the Report

This report presents the results of the work undertaken. The analysis seeks to capture costs and benefits in a comprehensive framework. However, as with all analyses of this nature, there are omissions, possibly significant ones, and uncertainties reflecting the lack of scientific knowledge and / or consensus around some of the effects for which valuations are sought. This, it should be added, applies to all the waste management options which the study addresses (and it does not address all treatments in a comprehensive manner).

The study follows the following lay-out.

Section 2: Potential Treatments for Biodegradable Municipal Wastes

Section 3: Current Situation Regarding Municipal Waste - Arisings, Composition And Treatments

Section 4: The Economic Analysis Of Options For Managing Biodegradable Municipal Waste – External Costs

Section 5: The Economic Analysis Of Options For Managing Biodegradable Municipal Waste – Financial Costs

Section 6: Results Of The Economic Analysis (in which the external and private cost analyses are combined)

Section 7: Future Scenarios (in which country-specific projections are made, and the private and external costs are estimated to the year 2010)

Section 8: Observations, Conclusions and Recommendations.

Further details are to be found in the supporting Appendices.

2.0 POTENTIAL TREATMENTS FOR BIODEGRADABLE MUNICIPAL WASTES

Municipal waste is treated in different ways in different countries in the EU and the Accession States. The variation across countries reflects a combination of:

- Differing levels of emphasis on source separation, enabling different approaches to treatment of waste; and

- Different approaches, relating to historical, economic, geological and cultural factors, to waste treatment (these approaches and the cultural factors being related, in a dialectical way, to the approaches to collection of waste – the two 'co-evolve').

As will become clear, some parts of Europe collect separately as much as 60% of all municipal waste (Flanders) whilst others carry out very little separation of wastes. As regards residual waste, some rely very heavily on incineration of household wastes (e.g. Denmark), whilst others landfill the majority of the municipal waste collected (e.g., Ireland, Italy, Spain, UK, Portugal, Accession States).

This Chapter reviews the different treatment options available for the treatment of municipal waste.

2.1 Landfill

The landfilling of waste has occurred for many years. All Member States and Accession States landfill some waste though several Member States are implementing, or have implemented restrictions or bans on the landfilling of municipal waste other than under specific conditions. In some countries, the majority of municipal waste is landfilled. The technical barriers can be said to be relatively few. However, it should be recognised that the term 'landfill' is used to refer to a wide range of facilities across Member States, from primitive dumps to sites which are engineered specifically for the purpose (and sometimes, for specific wastes), and frequently inspected. In some of the countries being examined, a significant quantity of municipal waste is landfilled in uncontrolled fashion in sites which are barely engineered, if at all.

The degradation of biodegradable wastes under landfill conditions creates methane. Methane is a powerful greenhouse gas (30 times or so more powerful than carbon dioxide) and the Landfill Directive is designed partly to address the issue of methane emissions from landfills. Notwithstanding the fact that inspections take place, and acknowledging the intentions to reduce impacts of landfilling, accidents do happen. Methane gas can build up in pockets and create explosions. For this reason, biological treatment to stabilise waste before landfilling is becoming an important pre-treatment for landfill in some countries. Furthermore, the land area occupied by landfills is considerable.

As long as landraises are not deemed acceptable to communities where they are proposed, the availability of landfill void space might be expected to be conditioned by demand for primary minerals and aggregates, which generates the void space that potentially becomes a landfill. Since many countries are seeking to encourage greater re-use of construction and demolition materials, and re-use of parts of buildings, it might be expected that where these initiatives are successful, the rate at which void space is created in the future will fall.

In addition, some countries use waste materials (including incinerator bottom ash, ash from other power stations and 'glasphalt', a product of recycled glass) to displace aggregates in construction and road-building projects. This will also slow down the rate at which void space is created. Since waste arisings are not falling, the conclusion that one might draw is that the supply of landfill void space will come under increasing pressure to meet demand under any scenario which represents 'business-as-usual' for wastes other than those used for construction purposes.

The business-as-usual scenario is increasingly difficult to define as the situation is changing. Quite apart from space, the principal influences on the degree to which landfill is used may be expected to be:

- Public opinion – landfills create significant disamenity effects (though these may fall over time as neighbours become accustomed to them); and
- Member State / Accession State legislation / plans.

Obviously, the Landfill Directive is a most important driver where the latter is concerned. Article 5 (2) sets out a schedule for Member States to reduce the amount of biodegradable municipal waste (BMW) landfilled. This has to be reduced in the following ways:

- By 2006, to 75% of the amount of BMW that was landfilled in 1995;
- By 2009, to 50% of the amount of BMW that was landfilled in 1995;
- By 2016, to 35% of the amount of BMW that was landfilled in 1995.

A 4 year derogation period exists for those Member States who were landfilling more than 80% of all municipal waste in 1995. This includes the following countries:

- Greece
- Ireland
- Italy
- Portugal
- Spain
- United Kingdom
- Cyprus
- Estonia
- Hungary
- Poland

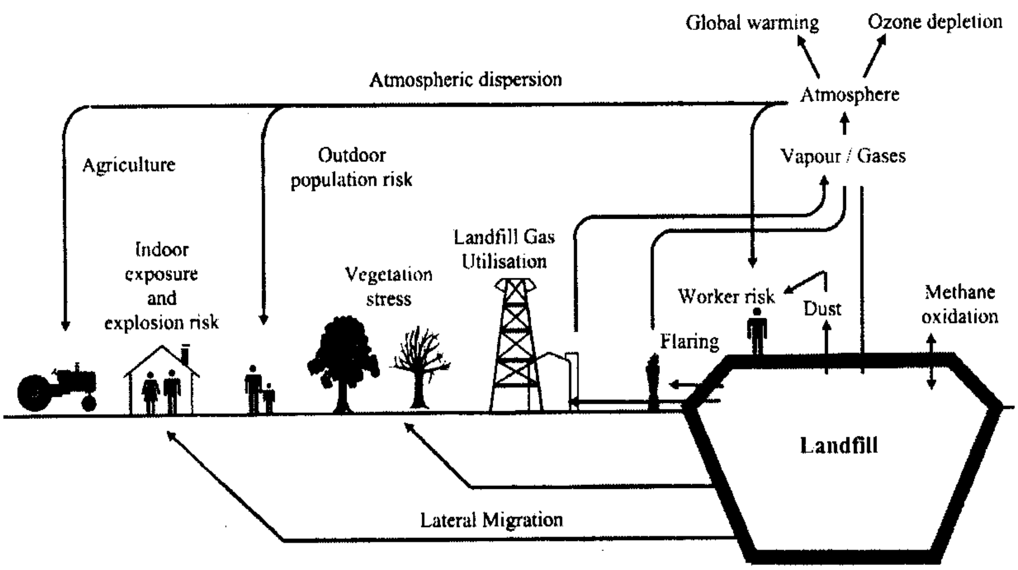
- Slovenia

All municipal wastes *can* be accepted by landfill. These wastes generate different emissions depending upon their potential to degrade under landfill conditions, and this affects the impacts of landfilling (see Figure 2). Different materials also degrade at different rates, and the contribution of different fractions to leachate will vary. Leachate will quite possibly affect groundwater at some later date. Whether, and if so, when leachate will become a problem will be determined in part by the landfill lining and the geological characteristics of the site.

The only 'end product' for landfills is landfill gas, which if collected can be used to generate energy. There will be markets for the energy, and some countries effectively support the generation of energy from landfill gas. The UK has done so explicitly under the Non-fossil Fuel Obligation (NFFO) and will do so implicitly (in future) through exemptions from the climate change levy (which will be introduced for other power sources in 2001).

The final residues in landfills consist of material which has not degraded (in landfill conditions) and the leachate residues which may be treated through various approaches. The former may have substantial carbon content. As such, to the extent that certain materials which might degrade under aerobic conditions do not do so in landfills, landfills may be considered to be a net sequester of carbon. Bramryd (1998) has likened them to a peat-bog for this reason.

Figure 2: Emissions to Air from Landfill and Exposure Pathways



Source: Gregory et al (1999)

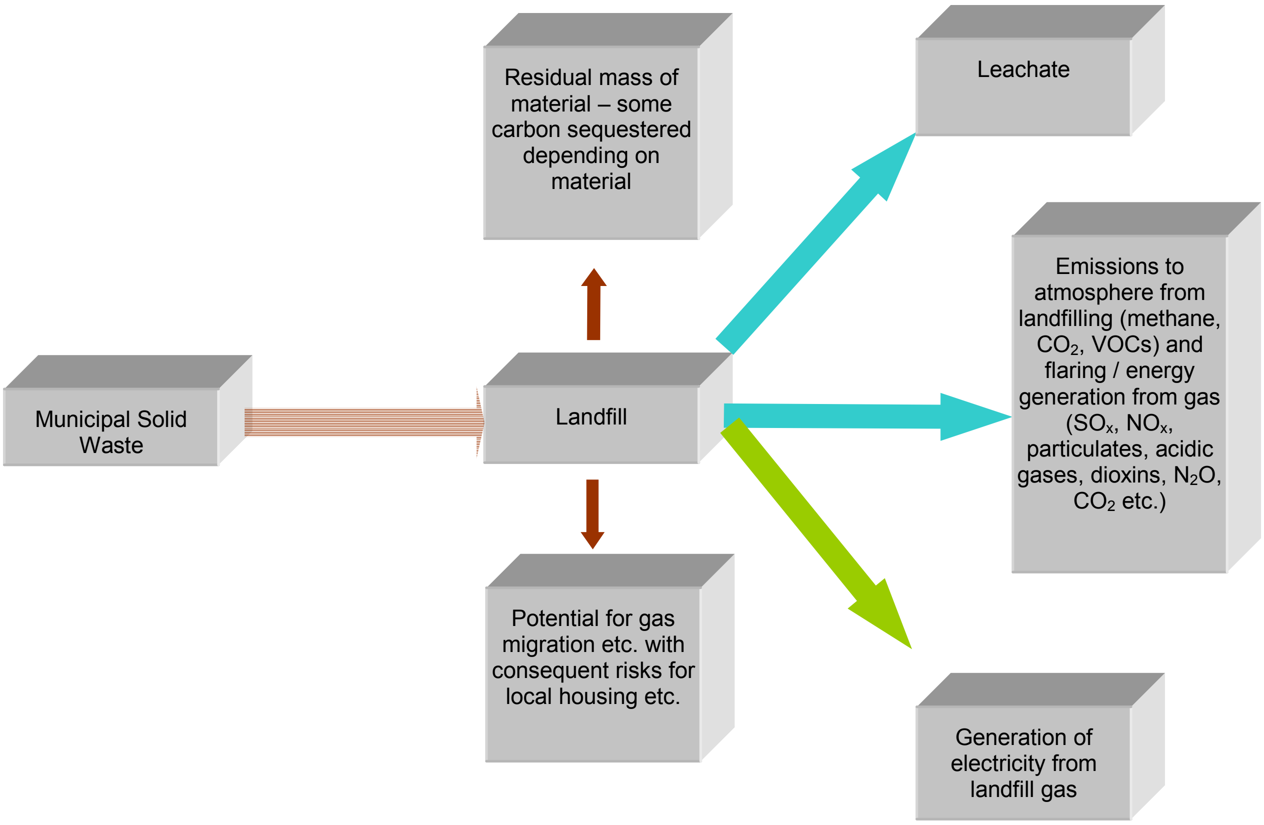
Two broad types of landfill strategies can be identified. Traditional landfills are uncontrolled and allow leachate to be released into the soil surrounding the landfill without restriction. This 'dilute and disperse' method is, however, no longer considered an appropriate operation method in view of the serious risk posed by leachate to groundwater supplies and the potential uncontrolled accumulation, and movement, of landfill gas. Most modern MSW landfills are therefore controlled and operated using the principle of 'containment'. Landfilled waste is separated from the environment by liners, and both leachate and landfill gas are collected and treated, including after the closure of the landfill.

Containment of waste combined with the operation of the landfill as a large 'bioreactor' has been proposed. This involves operating the landfill to accelerate the decomposition processes, such that the production of leachate and landfill gas occurs as early as possible and when the collection and treatment systems are in working order (Bramryd 1998).

Mechanical biological treatment (MBT) is a valuable tool for pre-treating wastes prior to landfilling. Such pre-treatment can lead to the material to be landfilled being relatively benign in respect of its potential to generate methane and leachate (MBT is examined below).

A schematic representation of the process is shown in Figure 3 below.

Figure 3: Schematic Representation of Landfill Inputs and Outputs



Note:
Red arrows represent residual materials
Blue Arrows represent 'negative outputs' (environmental costs)
Green Arrows represent 'positive' output (environmental benefits)

There are still uncertainties in the available knowledge as to how landfills affect human health. Recent work in the UK mentions the possibility (though no firm cause-effect relationship is suggested) of landfills being responsible for birth defects in the surrounding area (see Elliott et al 2001). Further work is being undertaken in this context.

2.2 Incineration (With Energy Recovery)

This section treats mass-burn incineration, fluidised bed incineration and incineration of refuse-derived fuel under the same heading. Of these different technologies, mass burn technology appears to be the most widely used.

In mass burn incinerators, waste is first fed into a feed chute where a ram pushes the waste on to the first section of the incinerator grate. The grate (rather like a downward escalator) may comprise a series of rocking sections (Rocker Grate); rotating rollers (Rolling Grate); or alternate fixed and moving sections (Reciprocating Grate). Each grate design aims to move the waste through the combustion chamber (furnace) with maximum exposure to oxygen at a high temperature. As the waste is propelled through the furnace, the carbonaceous/hydrogenous waste is dried and oxidised (combusted) with air supplied through the grate. The reaction leaves ash and flue gases to be quenched prior to cleaning and emission to the atmosphere. Energy recovery is obtained by the combustion gases transferring their heat to refractory-lined water tube sections as well as convective heat exchangers – both of which feed the boiler. Steam from the boiler can be used for district heating or in a turbine for power production to an electricity grid.

Refuse derived fuel (RDF) is manufactured by sorting wastes to remove wet putrescibles and heavy inerts (stones, glass, etc.) so as to leave combustible material. The remaining waste is then shredded and either burned directly, or pelletised prior to combustion (usually where the material is burned off-site, so that a densified fuel reduces transport costs). Manufacture of RDF is often an objective of MBT plants and the material may be incinerated in dedicated facilities, or co-incineration plants.

Fluidised bed incinerators operate with a bed of hot sand. The feedstock is prepared so that it is all of an equal size, sometimes using methods similar to that described above for RDF. The particles of sand and the feedstock are maintained under constant motion (fluidised) by a gaseous agent (air), which ensures good mixing of oxygen and the feedstock. The feedstock is maintained in the furnace until the carbonaceous and hydrogenous matter within the waste is oxidised (combusted), leaving ash and flue gases for cleaning and subsequent emission to the atmosphere. Variations on the basic design exist, but with all, either the sand never leaves the bed, or else it is re-circulated.

Incineration can, depending upon waste composition (which may exhibit seasonal variation), handle unsorted municipal wastes as well as wastes from which materials have already been separated. The different incineration technologies mentioned above may make more or less deliberate attempts to remove specific fractions of waste from the waste stream. For example, garden wastes may be best treated through composting both because of their seasonal nature, and due to the fact that much of the material (e.g. grass clippings) may have quite low calorific value.

An issue of significance for the operation of incineration plants is the calorific value of the input waste. Mass-burn grate incinerators tend to be designed for operation using material of a reasonably well-known calorific value. If the calorific value increases or decreases significantly, the input of waste to the plant has to reduced or increased to reflect the change. With fluctuations in the composition of wastes, the efficiency of the combustion process may change, altering the associated emissions. Where wastes reach very high calorific values, or where they are very wet, either changes in composition (separation at front-end, or mixing) may be required, or desirable. In extreme cases, the process itself may find the composition of waste difficult to cope with. This is generally believed to be less of a problem for fluidised bed incinerators.

One of the principal constraints on the use of incinerators is public opposition. In some countries, people simply do not want to live near these plants owing to problems of disamenity, and the emissions of NOx, SOx, HCl, particulates, heavy metals and dioxins associated with the plant. The first five of these are known to have effects upon human health.

For dioxins, the case is somewhat controversial. Draft reviews from the USEPA suggest the effects of dioxins may be worse than had originally been thought. It is important to note that the Incinerator Directive limit values refer only to the 17 chlorinated dioxins which are added to make up TEQs (toxic equivalents). Emissions of dioxin-like polychlorinated biphenyls (PCBs) are not included in any of the studies reviewing health impacts of incinerators. Also, Weber and Greim (1997) suggest that the similarity in action of chlorinated and brominated dibenzo-p-dioxins and dibenzofurans, would appear to imply that environmental and health assessments should be based on molar body burdens without discrimination of the nature of the halogen (characterizing the dioxin). This is not unimportant since there are about 5,100 halogenated dioxins, as well as polychlorinated dibenzothiophenes and thianthrenes, which are sulphur analogues of the dibenzodioxins and furans, and polychlorinated azobenzenes and azoxybenzenes (the list of potentially harmful chemicals is not a short one).

Hansen (2000), in a report from the Danish EPA, suggests that incinerators are a major source of dioxins in the country. Greenpeace Nordic (1999) suggest that the effect of controls of atmospheric emissions has been to shift dioxins away from flue gas emissions and into ash residues, which are frequently less well controlled than emissions to air. Furthermore, a study by De Fre and Wevers (1998) suggests that dioxin emissions to the atmosphere are underestimated at incinerator plants due to the way in which they are monitored (and this was recognised as a potentially important issue in the Hansen (2000) report).

Most mixed municipal wastes can be handled by incinerators as long as the constraints in respect of composition and calorific value are respected. Larger fractions may pose problems and may be inappropriate if they compromise the completeness of the combustion process.

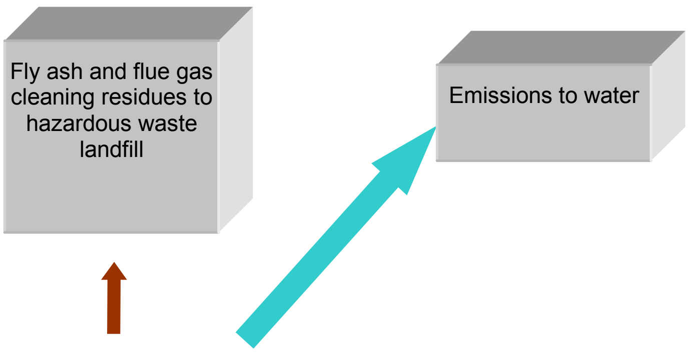
It is possible to extract metals, such as steel and aluminium, from the bottom ash. Indeed, this may be an advantage where wastes consist of mixed materials. However, the price paid for recovery of this material is usually far lower than in cases where the material has been source–separated since they are usually contaminated (being derived from the slag).

Some countries have given subsidies, or above-market prices for the energy generated by incinerators. This is usually mirrored in the local scenario for gate fees, as subsidies can sharply reduce the overall unit fee to be applied in order to offset managing costs. Italian provisions (the "CIP 6" Decree, then the "Green Certificates") and UK ones (Non-Fossil Fuel Obligation) are known to have affected the average gate fees significantly.[2]

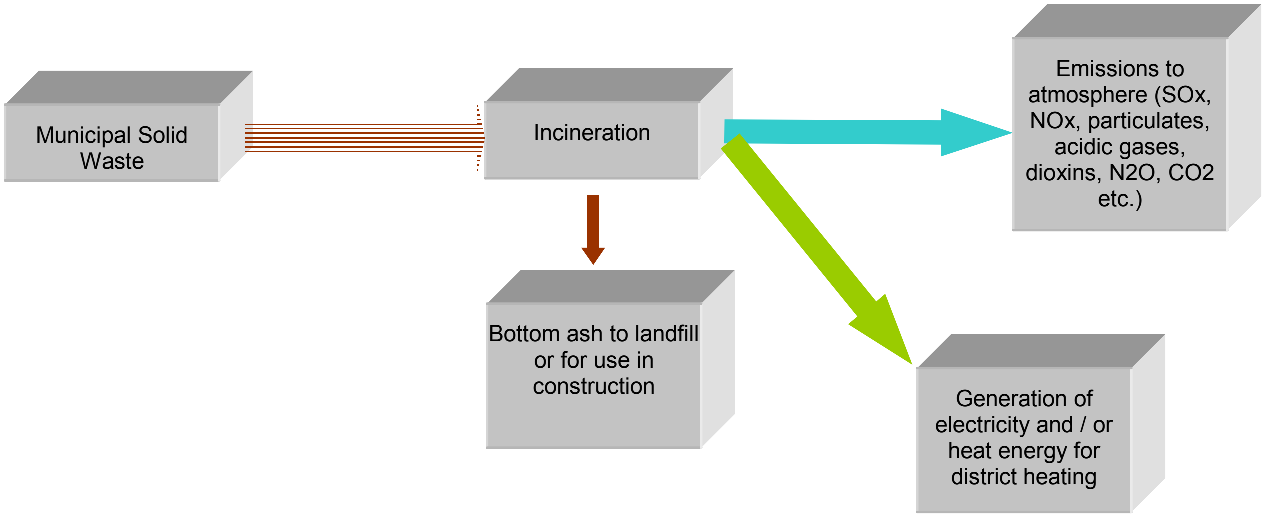
Final residues include bottom ash and fly ash, as well as waste waters. It is possible to use the bottom ash in construction applications, although some concerns remain as to the potential impact of this activity (if not now, then over the longer-term). Some countries also use fly-ash in construction post-stabilisation. Again, there are concerns that weathering will eventually lead to the release of what are often persistent and toxic chemicals contained in fly ash (and potentially, the presence of these will increase as requirements to clean flue gas become more strict). In th UK recently, concerns have arisen at plants where the mixing of bottom ash and fly ash has occurred prior to use of the material in construction applications.

Because various gaseous emissions from incinerators are known to have impacts upon human health (see above), a good deal of emphasis has been placed on flue gas cleaning. Depending upon the system used, a combination of solid and liquid residues will result from this process. These solid and liquid residues then have to be dealt with. In the case of fly ash, the toxic nature of residues requires careful handling and disposal to hazardous waste landfill facilities. There are likely to be important effects stemming from the Landfill Directive where disposal to hazardous waste landfill is concerned, though these will be especially significant where co-disposal is a common practice at present (this will have to cease). Fly ash generation tends to be greater at fluidised bed incinerators.

Figure 4: Schematic Representation of Incineration Inputs and Outputs



[2] In the UK, the NFFO scheme is effectively being replaced by a system of trading of Renewables Obligation Certificates. A statutory consultation document issued by UK Government on the Renewables Obligation suggests incineration will be excluded from this, but that gasification and pyrolysis will be included. The implication would be that the price support for electricity generated via incineration would be removed.



Note:
Red arrows represent residual materials
Blue Arrows represent 'negative outputs' (environmental costs)
Green Arrows represent 'positive' output (environmental benefits)

2.3 Pyrolysis / Gasification

Pyrolysis and gasification are relatively new methods for treatment of municipal solid waste and remain relatively unproven in European usage compared with classical moving grate methods. Although the technology is widely used and well established as an industrial process for energy recovery from hydrocarbons feedstock, their use as processes for dealing with heterogeneous, mixed municipal waste streams is at an early stage of development.

2.3.1 Pyrolysis

Pyrolysis is a process which transforms waste into a medium calorific gas, liquid and a char fraction in the absence of oxygen, through the combination of thermo-cracking and condensation reactions.

Pyrolysis involves indirect heating of carbon rich material. The aim is to achieve thermal degradation of the material at a temperature of some 500°C (a range 450-600°C is observable) in the absence of oxygen and under pressure. The temperature is usually maintained through indirect heating.

Suitable feedstocks that can be treated by a pyrolysis facility include sewage sludge, agricultural wastes, mixed organic waste including food waste, garden waste, paper pulp and pre-separated residual waste.

Pyrolysis produces gas, liquid and solid char. Specifically:

- Gas stream (uncondensed gases from pyrolysis), containing CO, CO_2, H_2, CH_4, C_2H_6, C_2H_4;
- Tar/oil (condensed gases from pyrolysis) - acetic acid, acetone, methane; and

- Char - pure carbon, with other inert material and heavy metals.

The relative proportions of these products depends on the type of pyrolysis employed and the reaction parameters. The process may be followed by a combustion step and/or extraction of pyrolytic oil.

There are three main types of pyrolysis: (i) Slow pyrolysis or carbonisation, (ii) conventional pyrolysis and (iii) fast/flash pyrolysis (further separated into vacuum and fluidised bed). The cost of pyrolysis depends on the technology employed and in general it can be said to vary from medium to high. When compared with anaerobic digestion the cost is similar, but it is typically higher than the cost of an incineration facility. On the other hand, the plant scale is usually much smaller and it might be argued that if incinerators were constructed at the same scale, they would have comparable or even higher costs (as the diseconomies of reduced scale are considerable). Hence, such facilities may be better-suited than incineration to scenarios where residual waste is mechanically separated into a smaller fraction for pyrolysis.

In fast pyrolysis, the carbonaceous material is fed into a chamber and rapidly heated to medium temperatures (400-700^{o}C) such that it reaches vapour phase almost instantly. It is then extracted from the chamber and quenched (cooled) rapidly. The very fast heating/rapid quenching means that few of the long carbon-hydrogen chains are broken and so the vapours condense into a liquid fuel (bio-oil), as opposed to a gas, as would be the case for slower heating and quenching.

In vacuum fast pyrolysis, the feedstock is introduced to the reactor in a vacuum, where it is pyrolysed on a transported bed at 450^{o}C. Some gas is evolved and provides heat for the process. The vacuum technique has been developed primarily for the recycling of chemicals from tyres to the chemicals industry, since large quantities of pyrolytic oils can be obtained. Fast pyrolysis usually occurs in a reducing atmosphere, with a complete absence of oxygen to avoid widespread gasification, thereby maximising bio-oil recovery.

The chemical reactions involved are influenced by three factors:

- Input materials (chemical composition, water content)
- Reactor design (vertical shaft or batch reactor, rotating tubular or fluidised bed reactors, under vacuum or controlled atmosphere)
- Operating conditions (temperature, pressure, reaction time).

High reaction rates minimise the formation of char, to maximise bio-oil production (up to 80% mass yield of bio-oil). The degree of ‘vacuum’ is a key influence on the quantities produced. Higher temperatures used lead to greater gas formation. Energy for reaction can be directly or indirectly provided.

The fact that the gaseous, liquid and solid fractions are relatively homogeneous makes thermal valorization less problematic (at least in theory) than in the case of incineration. Residence times in the furnace vary, usually being shorter for rotating kiln or fluidised bed designs and longer for fixed bed designs.

The char, which is evacuated through an airlock system, is typically screened for various metal fractions. There is usually some contamination with heavy metals. Following screening, in integrated processes, the fuel may be sent to a combustion or gasification unit. Alternatively, it may be washed. If the pyrolysis process has incorporated liming in the process, this can lead to leaching out of chlorine as calcium chloride, enabling the remaining fine coke residue to be used as auxiliary fuel in industrial applications, such as coal-fired power plants and in cement rotary kilns. Where the desired output is chemicals for synthesis, a second gasification process is used to generate syngas, which is converted chemically into methyl alcohol or ethanol. Generally the synthesis gas can be used either to substitute natural gas or to generate electricity.

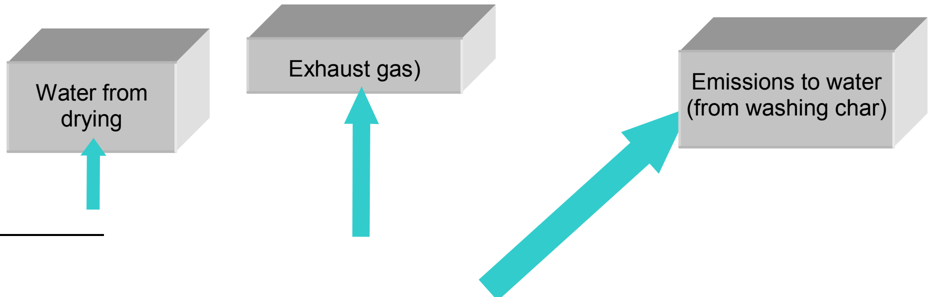
The recoverable energy content varies from 200 to 400 kWh/t of waste, although reports differ regarding these claims. Energy production and greenhouse gas production are lowered due to the starved air conditions. Less volatile heavy metal species remain in char while volatile species need to be captured by gas cleaning systems and treated as hazardous materials.

Regarding the flue gases and wastewater generated by the facility these can be treated by means of common pollution prevention technologies. The heat from the flue gas can be recovered by means of a boiler with a super heater and economiser for the generation of high-pressure steam.

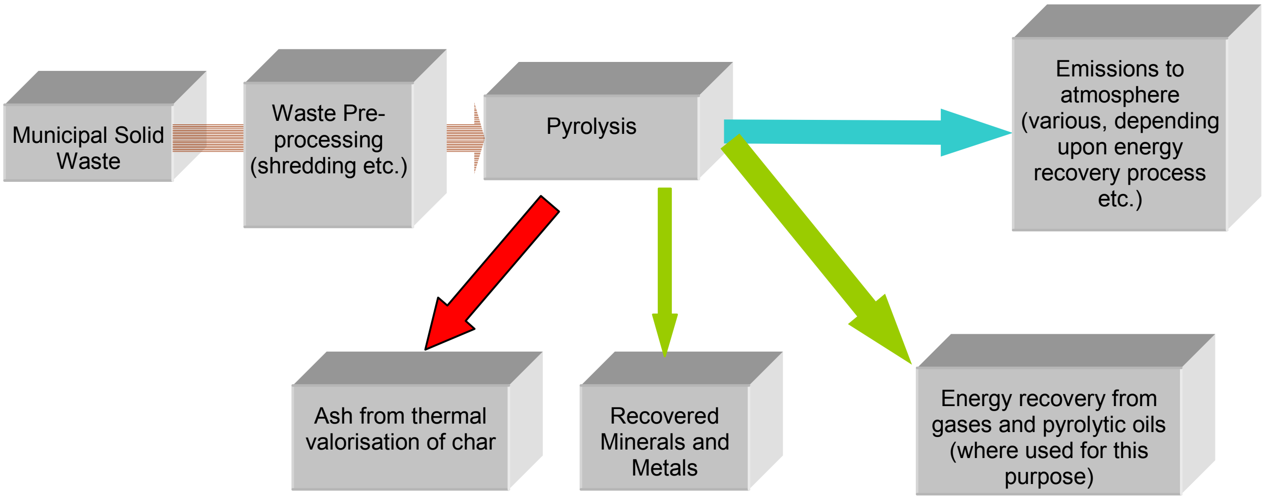
Major technical issues with pyrolysis include: heat transfer to the waste material; precise process control to achieve the desired mix and yield of products; and product separation and collection, especially of bio-oil, which needs to be condensed. Process energy is self-propagating. Pyrolysis tends not to be an efficient energy conversion technology since much of the fuel produced is consumed within the operation.

For municipal waste, it would appear that the major technical problems relate to the input materials. It is generally accepted that these have to be relatively homogeneous in order for the process to function without problems. For this reason, plants tend to be equipped with front-end equipment designed to transform the waste through pre-processing to ensure the proper operation of the facility (such as a shear shredder to adjust the particles size of the feedstock). Equally, pyrolysis may be a suitable process for treating the output of mechanical biological treatment plants.

Figure 5: Schematic Representation of Single Pyrolysis Process Inputs and Outputs[3]



[3] Note, the variety of designs makes it difficult to characterise this as one process. See Fontana and Jung (2001).



Note:
Red arrows represent residual materials
Blue Arrows represent 'negative outputs' (environmental costs)
Green Arrows represent 'positive' output (environmental benefits)

2.3.2 Gasification

Gasification involves heating carbon rich waste in an atmosphere with slightly reduced oxygen concentration. The majority of carbon is converted to a gaseous form leaving an inert residue from break down of organic molecules.

Gasification is a thermo chemical process involving several steps. First, carbonaceous material is dried to evaporate moisture. Depending on the process, pyrolysis then takes place in a controlled, low air environment in a primary chamber, at around 450^{o}C, converting the feedstock into gas, vapourised liquids and a solid char residue. Finally gasification occurs, in a secondary chamber at between 700-1000^{o}C (dependent on gasification reactor type). Here the pyrolysis gases and liquids and solid char undergo partial oxidation into a gaseous fuel, comprising a variety of gases (dependent on reactor configuration and oxidant used). These gases include carbon monoxide, carbon dioxide, hydrogen, water, and methane (and much smaller concentrations of larger hydrocarbon molecules, such as ethane/ethene). Oils, ash tars and small char particles are also formed in the reaction, acting as contaminants. The heat source for the gasification process can be heated coke. Superheated steam can also be injected at this point to facilitate the conversion into gaseous fuel.

Process description varies for different specific technologies and is generally patented. The conversion process can utilise air, oxygen, steam or a combination of these gases. Gasification using air – the most widely used technique – produces a fuel gas suitable for boiler/engine use, but it is difficult to transport in pipelines. Nitrogen is evolved since air is used in the oxidation process.

Gasification using oxygen (which is more expensive due to cost/hazard of oxygen generation) produces a medium heating value (MHV) gas which can either be used as a synthesis gas (e.g. for conversion to methanol) or for limited pipeline distribution. Steam (or pyrolytic) gasification produces a MHV gas.

A variety of gasification reactors (running at either atmospheric pressure or pressurised) have been developed, including fluidised and fixed bed. There are numerous advantages/disadvantages to each configuration. Incomplete oxidation due to reactor design and feedstock anomalies can contaminate the product gas, and where air is used, this will result in higher than expected NOx emissions. Circulating fluidised bed gasifiers are seen as more versatile since char can be recycled.

The fuel gas can be used in thermal combustion engines to produce energy; in a steam turbine or a boiler; or as a raw material resource to produce methanol, hydrogen or methylacid. Syngas includes carbon dioxide, methane, carbon monoxide, hydrogen, nitrogen and ammonia. Small quantities of hydrochloric acid, hydrofluoric acid, hydrobaric acid, sulphur dioxide and nitrogen oxides and particulates are produced along with trace metals or heavy metals, notably cadmium and mercury.

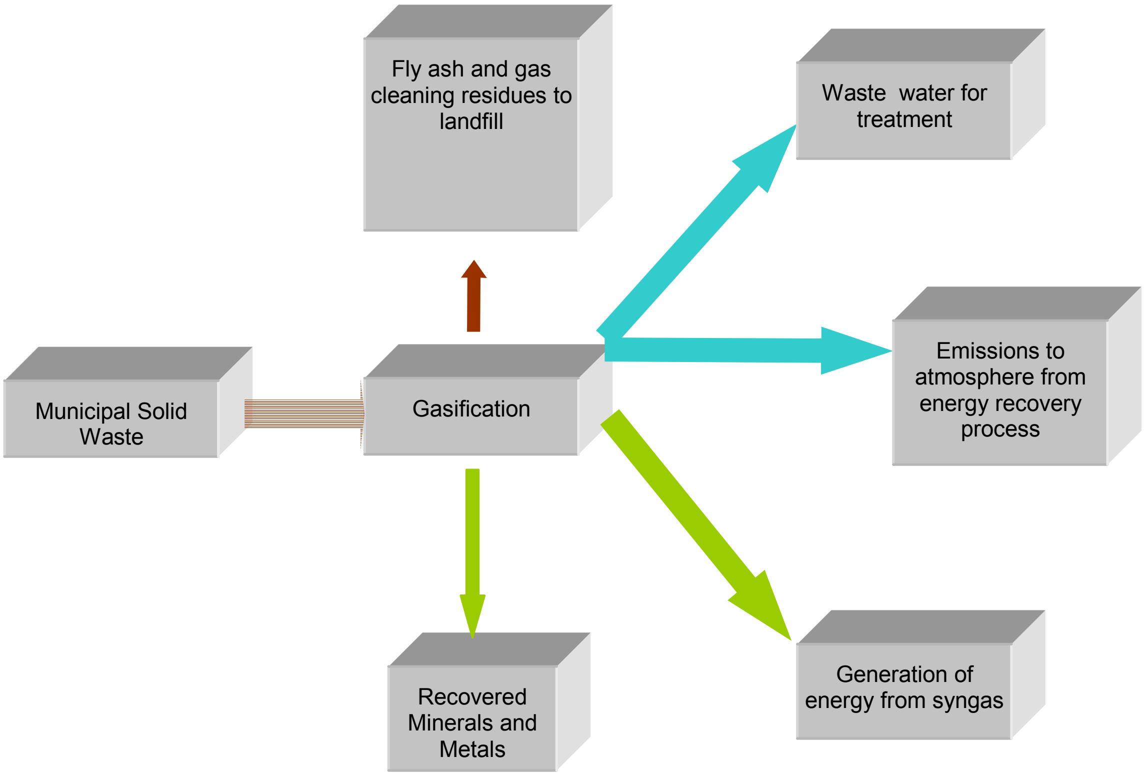
Gasification is widely considered as an energy efficient technique for reducing the volume of solid waste and for recovering energy. Useable energy of some 500 to 600 kWh per tonne of waste is generated by gasification.

Gasification technologies have been operated for over a century for coal producing “town gas” and have long been promoted as being a viable, cleaner alternative to incineration for residual municipal wastes.

Gasification is more widely used and more developed than pyrolysis for several reasons. First, a highly efficient process produces a single gaseous product. Second, gasification does not have the heat transfer problems associated with pyrolysis. However, plants are known to have closed down due to waste variability and material handling problems. Newer processes have been developed in order to overcome these problems through extensive pre-processing of the feedstock waste.

A number of Gasification and Pyrolysis processes are at commercial scale at the moment, applying a number of combinations of different techniques such as pyrolysis, combustion and gasification. According to a survey carried out in 1997 [Juniper, 1997] there were 16 technologies at varying stages of development with Siemens, Thermoselect and Von Roll being the most advanced European technologies with the first commercial plants in various stages of completion by that time. The Thermoselect plant at Karlsruhe, however, recently suffered problems associated with heavy metal emissions. Siemens has also effectively withdrawn from this market, having had problems with carbon monoxide emissions at a plant in Furth.

Figure 6: Schematic Representation of Gasification Inputs and Outputs



Note:
Red arrows represent residual materials
Blue Arrows represent 'negative outputs' (environmental costs)
Green Arrows represent 'positive' output (environmental benefits)

2.4 Mechanical Biological Treatment

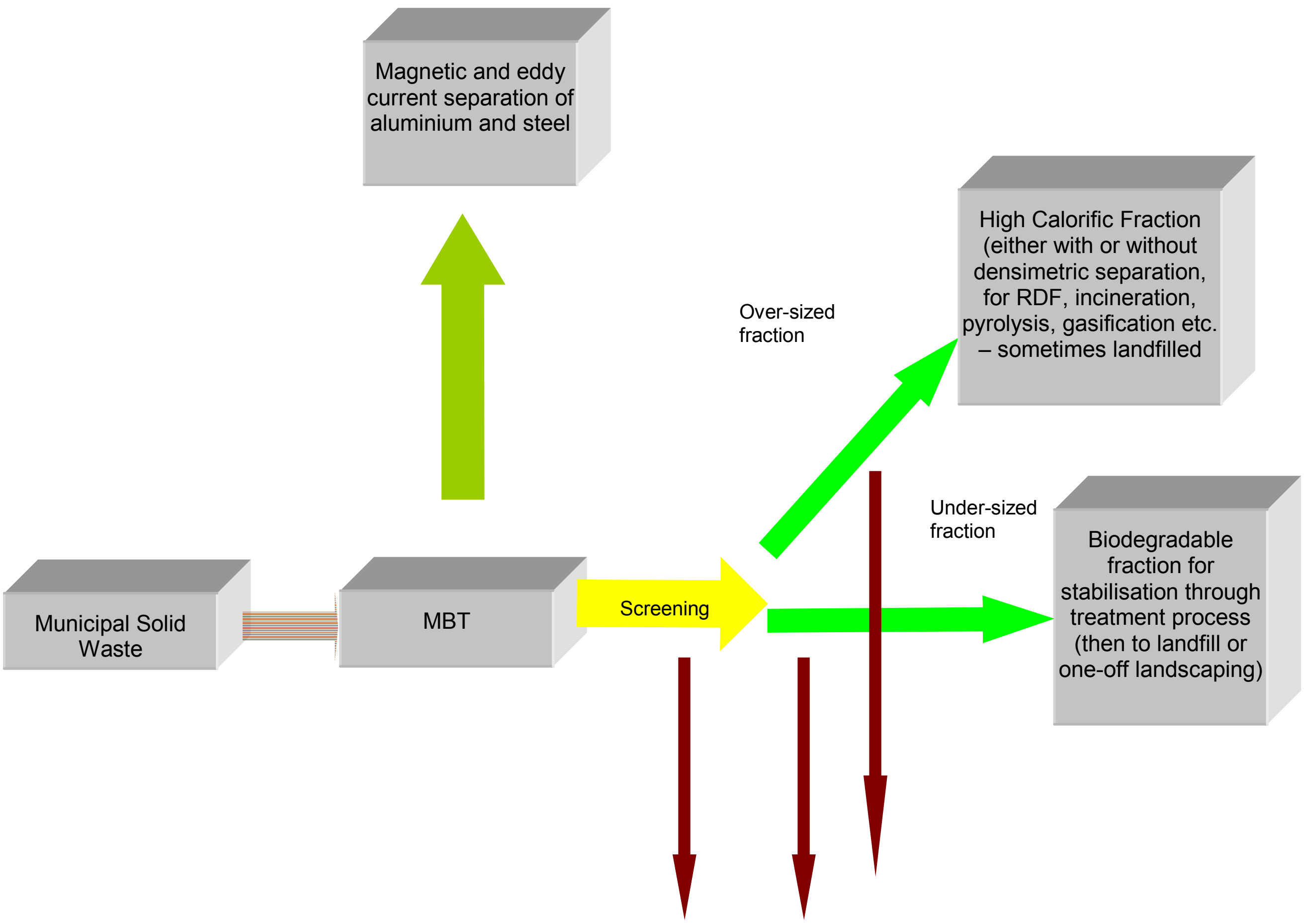
Mechanical biological treatment is a process designed to optimise the use of resources remaining in residual waste. Usually, it is designed to recover materials for one or more purpose, and to stabilise the organic fraction of residual waste. The benefits of this process are that materials and energy may be recovered, void space requirements are reduced and gas and leachate emissions from landfill are significantly reduced.

The mechanical treatment phase involves segregation and conditioning of wastes. The process involves primarily the shredding / crushing and screening of materials so as to:

- Open waste bags (where necessary);
- Extract undesirable components that may obstruct subsequent processing;
- Optimise particle size for subsequent processing;
- Segregate biodegradable materials in the underflows of primary screening, to be sent to the biological treatment process;
- Segregate materials with high calorific value, such as textiles, paper and plastics, in the overflows of primary screening, to be sent for RDF production. Also, segregate those suitable for further material recovery or to be landfilled; and
- Homogenise materials destined for biological treatment.

The type of shredding and crushing machinery to be used will be determined by the materials to be handled, the objectives of the treatment and the required processing capacity. One type of plant design is shown in Figure 7.

Figure 7: Schematic Representation of Mechanical Biological Treatment Inputs and Outputs



Note:
Red arrows represent residual materials
Blue Arrows represent 'negative outputs' (environmental costs)
Green Arrows represent 'positive' output (environmental benefits)

The permutations regarding the design of the plant are many and varied (see Zeschmar-Lahl et al 2000). In principle, however, all materials can be accepted at such plant, the intention often being, with some of them, to pre-treat / separate prior to landfill / thermal treatment / recycling / recovering specific fractions. In Figure 7 above, it is assumed that the plant is designed to separate a biodegradable fraction from residual waste for biological treatment prior to landfilling, or perhaps, one-off landscaping applications ("grey compost" or "stabilised biodegradable waste"). Apart from this, the plant may include equipment for metals recovery, extraction of mineral fractions, and for partitioning of high calorific fractions which could be sent for thermal treatment (sometimes through manufacture of RDF). Inert fractions may be landfilled / recycled as appropriate.

2.5 Composting

Composting is the biodegredation of organic matter through a self heating, solid phase, aerobic process. This converts organic matter into a stable humic substance. The microorganisms that carry out this process fall into three groups; bacteria, fungi and actinomycetes. While there are no strictly defined boundaries, the biological activity can be seen in three stages:

- Stage one is the consumption of easily available sugars by bacteria which causes a rapid rise in temperature.
- Stage two involves the break-down of cellulose by bacteria and actinomycetes; and
- Stage three concerns the break-down of the tougher lignins by fungi as the compost cools.

For this to take place efficiently, five key factors need to be considered; temperature, air supply, moisture content, the porosity of the material and its carbon to nitrogen ratio.

Figure 9 below illustrates the emissions from the composting process. These include:

- emissions to air, including:
    - o gaseous emissions such as carbon dioxide, by far the most prevalent gas released in the process, ammonia, methane, and some VOCs (some of which may be derived from biofilters using woody materials);
    - o bioaerosols, usually most prevalent when materials are being turned;
    - o odours (though these can be controlled in enclosed processes through use of biofilters); and

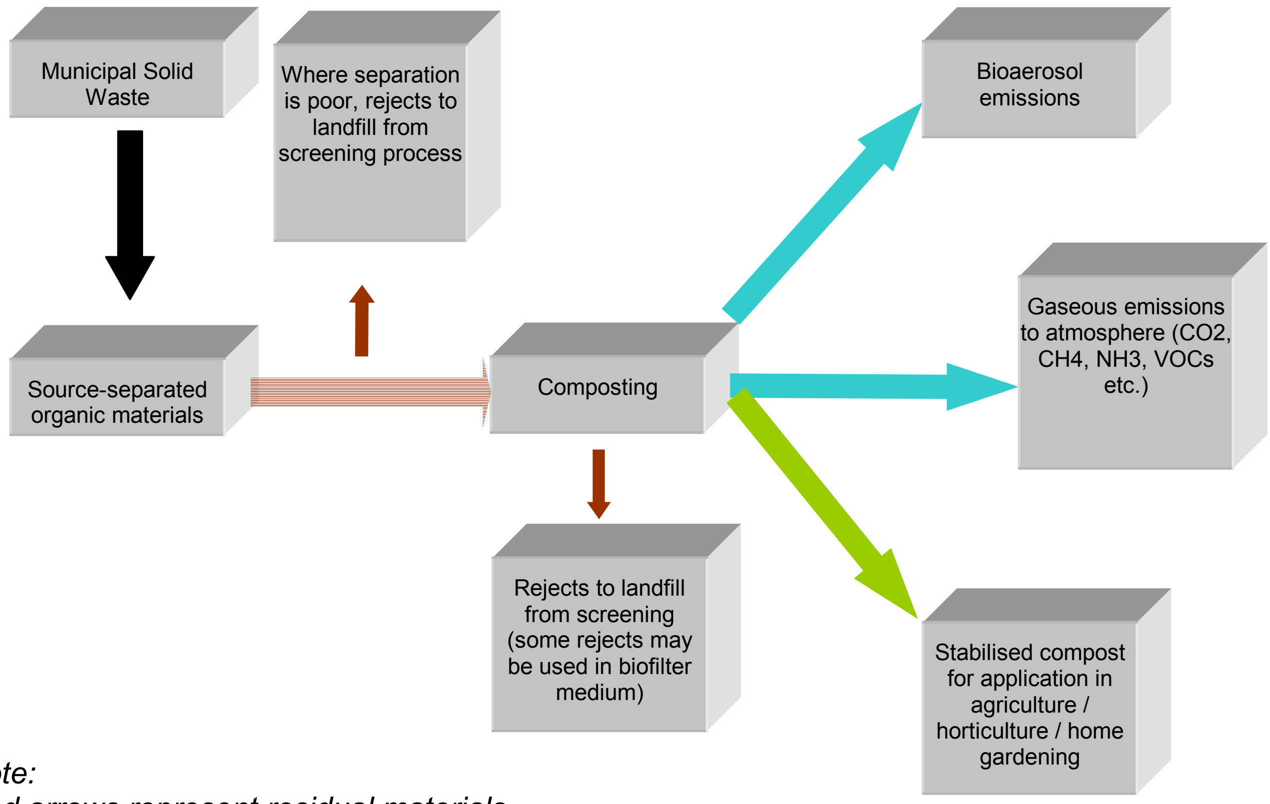
  - dust

- emissions to land, related to reject fractions (which can be minimised through source separation). To the extent that these are not biodegradable, or have been stabilised, they are less likely to give rise to problematic emissions if landfilled (where the process deals with source-separated waste); and
- in open processes, where no controls exist, leachate.

Probably the most problematic of these are issues associated with odour, and potentially, though clear relationships are difficult to establish, bioaerosols.

European policy exhibits a trend towards the development of source separated waste collection and composting and the promotion of home composting. Although the popularity of mixed waste composting is declining, it is carried out in France, Greece, Spain, and Portugal, whilst in Italy, Germany, Austria, and other countries, it is being progressively or totally 'converted' to MBT of residual waste. Several countries in the EU advocate the collection of source separated waste using a variety of collection and composting methods.

Figure 9: Schematic Representation of Composting Inputs and Outputs



Note:
Red arrows represent residual materials
Blue Arrows represent 'negative outputs' (environmental costs)
Green Arrows represent 'positive' output (environmental benefits)

The open air windrow system is a low technology system requiring minimum investment in terms of equipment and finance. Raw materials are heaped into long, low piles during the active composting phase before being set aside and left for a further period to mature.

Open air windrow composting is being progressively abandoned (increasingly used for garden waste only), and the trend is to compost food waste and other fermentable feedstocks using high technology systems which compost in-vessel, at least during early process stages. In many countries small facilities are exempted from provisions on enclosed composting of fermentable materials. The Austrian situation is interesting from such a standpoint since a high percentage of the overall composting capacity is actually covered through *"Bäuerliche Kompostierung"* (on-farm composting) managed by farmers in simplified rural sites. This also is being widely developed in Northern Italian Alpine Regions.

In addition to open air windrowing there are a wide range of technologies available for producing compost. These include reactor systems, tunnels and static piles. The more intensive technologies are used especially to compost so-called problem wastes such as sewage sludge and food waste.[4] Greater control can be maintained over the process so any pathogens present are more likely to be destroyed. The initial degradative phase can also be completed in a shorter time than with windrow systems, which is valuable if rapid processing of material is required.

All the systems require the material to be left for some time in piles for maturation, or curing. The duration of this phase varies widely with technology and end-product requirements, with potting applications requiring a very high maturation degree (usually achievable through a 90-120 days overall processing time).

2.5.1 Technical Problems / Primary Constraints

There are few major problems with composting processes as long as material is well aerated so that the process does not become an anaerobic one as opposed to one of aerobic composting. This is relatively straightforward to ensure. The degree to which aeration and / or turning occurs also has implications for the requirements in terms of structural stability of the input material. Table 1 illustrates some of these basic points

Table 1. Categorisation of Composting Methods

Method	Principles for feeding and turning	Demands on structure-stability of final input material	Type of Facility	Odour Control Possibilities (on scale of 1-4: 1 = poor, 4 = excellent)
Without forced aeration	Batch-wise and Static	Very high	Mattress/bed	1
		High	Windrow	1
With forced aeration	Batch-wise and static	High	Aerated-windrow	2
		High	Semi-permeable cover	3
		High	Container/box/tunnel-static	4
	Continuously and agitated	Medium	Indoor-mattress/agitated-bed	4
		Medium	Channel/agitated-bay	4
		Medium	Tunnel-agitated	4
		Medium	Tower multi-floor	4
		Low	Drum	4

Source: Adapted from Amlinger (2000)

There is increasing attention being paid to ensuring that processes achieve satisfactory levels of pathogen reduction, principally through ensuring the material reaches specific temperatures for minimum periods of time. This also implies the desirability of some form of monitoring.

[4] The importance of dealing with sewage sludge has increased since the banning of disposal of sewage sludge to sea waters.

Other constraints may be linked to issues of siting compost facilities, which may be problematic because of odour-related issues (see Table 1 above). Lastly, a constraint has been, historically, the nature of the materials composted. Impurities in the feedstock lead to poor quality products. Such products can undermine the markets for quality composts. In this context, standards act as a means of maintaining markets for quality composts and their absence makes it less likely that such markets can be reliably developed.

2.5.2 Type and Quality of Waste Appropriate to the Option

In the quest to produce quality composts, the process is largely confined to source separated fractions of biodegradable fractions. However, it is important to appreciate that, as far as municipal wastes are concerned, paper and card can also be composted as can some textiles. All kitchen wastes and garden wastes can be composted. To some extent, which wastes one chooses to collect for composting has to be related to the location and type of plant to which the materials are to be sent.

2.5.3 Characteristics and Quality of End Products and Potential Markets

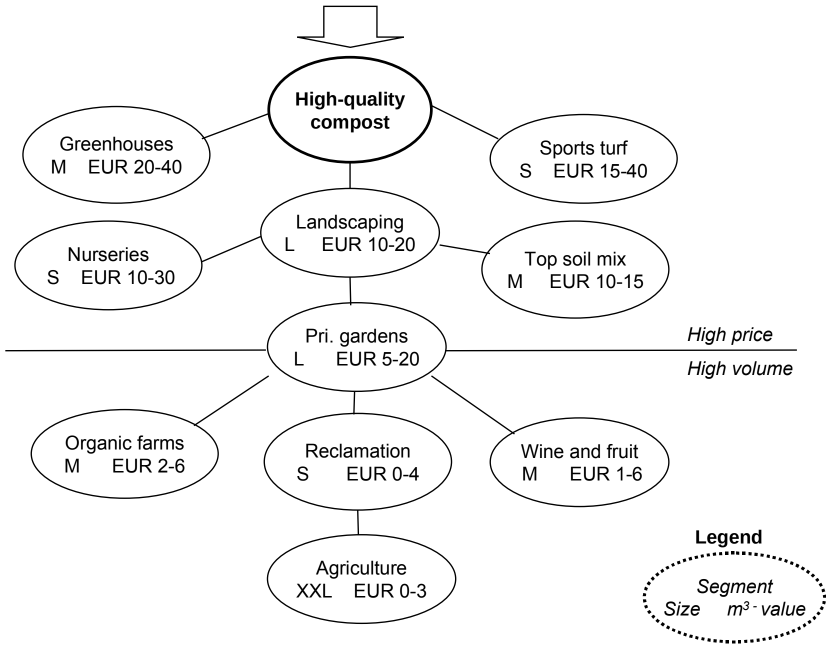
The characteristics and quality of the end products are very much determined by:

- The nature of the feedstock (how pure);
- The nature of the feedstock (its composition); and
- The extent of product maturation.

These, as well as process control, will determine product quality. This in turn determines the potential market for the product. Figure 10 shows typical prices and the size of different market outlets for composts in the EU.

Products made from cleaner feedstocks and with longer maturation times are more likely to reach quality markets (such as horticultural markets), but the nature of feedstock (for example, the relative mix of kitchen and garden waste) will also determine the nature of the product. ‘Fresher’ products (with shorter maturation times) are more likely to be appropriate for agricultural purposes.

Figure 10: Compost Marketing Hierarchy Indicating Market Prices and Volumes



Note: The volume is indicated as the relative size (small (S) to extra-extra-large (XXL)) of the market segment. Prices are known ranges for compost products within the market segment (EUR/m3). M. Carlsbæk, SOLUM (personal communication), in Amlinger (2000)

2.6 Anaerobic Digestion

Anaerobic digestion (AD) is the bacterial decomposition of organic material in the (relative) absence of oxygen. The by-products of this process include biogas (comprising principally carbon dioxide and methane, which is capable of combustion to generate energy), as well as a semi-solid residue, referred to as a digestate. With further treatment – normally through composting – the digestate from source-separated biowastes may be used for agricultural/horticultural purposes. Some countries allow direct application of the digestate onto farmlands (e.g. Sweden, Denmark).

The high degree of flexibility associated with AD is claimed to be one of the most important advantages of the method, since it can treat several types of waste, ranging from wet to dry and from clean organics to grey waste. The suitability of the method for very wet materials, for instance, has been addressed as an important feature in those scenarios where source separated food waste cannot be mixed up with enough quantities of bulking agents such as yard waste (namely, many metropolitan districts). AD of MSW has been commercially available for approximately 10 years and in that time, the heterogeneous and variable nature of the feedstock has given rise to a considerable number of different processes in operation in many different countries.

This study is concerned primarily with the digestion of derivatives of municipal wastes (specifically source-separated biowaste because of the associated benefits examined below). The inclusion of other feedstocks, such as sewage sludge, alters the quantitative aspects of digestate. However, it is important to note that the mixing of household waste with these feedstocks may improve both the environmental and economic aspects of the process and has been adopted in a number of plants (particularly, co-digestion with slurries and manure at small-scale farm based plants) and may be more adopted widely in the future. For example, the addition of sewage to the organic fraction of MSW will increase the nutrient level as well as adding moisture content. The heavy metal concentration in sludge should be carefully addressed as the tight limit values for quality composted products which exist in some countries might be difficult to meet where sludge is used.

In Germany, there is a situation in which residual waste, consisting mainly of food residues and non-recyclable paper, goes through a sieving process (a form of mechanical biological treatment) and the undersize fraction is fed to the digester. Digestion of mixed or residual waste is adopted also in other Member States such as France, Italy, and increasingly in Spain. Hence, the term anaerobic digestion can be used to cover the range of processes covering those that occur in bioreactor cells/landfills to the digestion of source separated materials.

Anaerobic digestion generally involves three stages:

- pre treatment;
- anaerobic digestion; and

- post-treatment.

2.6.1 Pre-treatment

In general, it is accepted, just as with composting, that source separated MSW makes materials handling much easier. Even source-separated MSW will, however, require further separation to remove wrongly sorted materials such as plastics, metals and oversized components.[5] Separation can be carried out under wet or dry conditions. Following this, a process of size reduction is used to create a more homogenous material which will aid fermentation and facilitate processing. The size-reduction can use screw-cutting, milling, drumming, pulping or shredding machines.

2.6.2 Digestion

Just as in the case of composting, there are a number of different techniques falling under the definition of anaerobic digestion (AD). They are usually distinguished on the basis of operating temperature (thermophilic plants operate at around 55^{o}C and mesophilic at around 35 ^{o}C) and the percentage of dry matter in the feedstock (dry systems with more than 20% dry matter, wet systems have less than 20% dry matter).

As regards the AD of MSW in-vessel, technologies include

- wet single-step, in which MSW is slurried with process water to provide a diluted feedstock for feeding in to a mix tank digester. the process can be used for MSW on its own, but the wet process lends itself to co-digestion with diluted feedstocks such as animal manure and organic industrial wastes;
- wet multi-step, in which MSW is again slurried and fermented by hydrolytic and fermentative bacteria to release volatile fatty acids which are then converted to biogas in a high-rate industrial wastewater anaerobic digester. The system lends itself to digestion of MSW and wet organic waste from food processors;
- dry continuous, in which a digestion vessel is continuously fed with a material with 20-40 percent dry matter through batch loading. In both mixed and plug-flow variants, the heat balance is favourable to thermophilic digestion;
- dry batch, in which a batch is inoculated with digestate from another reactor and left to digest naturally. Leachate is re-circulated to maintain moisture content and redistribute methane bacteria throughout the vessel;
- sequencing batch, essentially a variant of the dry batch process, in which leachate is exchanged between established and new batches to facilitate start up, inoculation and removal of volatile materials from the active reactor. After digestion becomes established, the digester is uncoupled from the established batch and coupled to a new batch in another vessel.[6]

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[5] Note that this is a consequence of the technical features of the AD process which requires coarse inerts to be removed. This is not always necessary at composting plants where separation achieves a purity in excess of 95% or so.

[6] The concept of bioreactor landfills is not discussed here.

As will be clear from the aforementioned discussion, the picture regarding AD (and indeed composting too) as applied to MSW is complicated slightly by the fact that there may be advantages in some circumstances to treating MSW in conjunction with other wastes such as organic industrial wastes, food processing wastes and agricultural wastes. As such, there may be specific localities where co-digestion is more feasible than in others.

2.6.3 Post-treatment Processes

After digestion, if the feedstock is wet, the material may be spread directly to land (especially where co-digestion with other wastes has occurred). Usually, this requires licensing as digestate is often considered as a “sludge”. Sometimes (e.g., in Denmark) it is considered as a product and can thus be applied onto farmlands with no licensing procedure.

Alternatively, solid and liquid fractions can be separated in which case, after two to four weeks’ maturation (sometimes longer depending upon the application), a fully stabilised compost will have been developed. The liquid fraction may either be recycled for dilution of fresh waste, applied to land as a liquid fertiliser (again, frequently under licensing), or sent to a wastewater treatment plant (often following some separation of solids). The possibilities for spreading of liquid wastes on land are likely to be regulated in different Member States.

2.6.4 Technical Problems / Primary Constraints

Whilst it is recognised that a there is a lot of potential in AD, at present a number of barriers exist to its widespread adoption. In spite of an increasing amount of information being published on AD, it still remains relatively scarce. There is a need for definitive information for the different processes, both regarding economics and the environment, as the available information gives conflicting messages regarding environmental impacts, commercial viability, etc. In addition, some of the information is based on theoretical studies, or on commercial literature, highlighting the need for independent data from monitoring real plants.

There are a number of areas of risk associated with AD at present:

- The main area of concern in AD is the guarantee of long term performance of a plant which is key to its economical feasibility. This risk can be reduced through technological developments but the associated costs may affect the economics in the short term. More plants being built in the future will enable learning to occur and this may increase confidence;
- As use of the technology increases, there is a risk that plants will be competing for the ‘best’ feedstocks;
- Specific investment costs are generally much higher than with composting itself. These could be partially offset by subsidies to the energy produced, though these should clearly be introduced with care and not with the explict objective of ‘making AD commercially viable’ in mind;

- A close integration between waste management and water management would be helpful to further development. This would reduce the extra costs related to discharge of excess waters from AD to a waste-water treatment plant. Such a condition occurs only rarely across Europe, most often where water utilities are involved in the process;
- Finally, more plant failures would reinforce the technology's relatively poor reputation at present. This has been an issue recently in Denmark due to a number of technical problems.

As mentioned above, the financial aspect is an important issue. Because the treatment method is both relatively rare at present (it is only part of waste management strategies in three countries, Germany, Austria, Belgium and Denmark, with some application on mixed or residual waste in France, Spain and Italy, though a small-scale plant is in operation in the UK), there is still a good deal of uncertainty as to its viability. Poor performance in the past has not helped matters.

2.6.5 Type and Quality of Waste Appropriate to the Option

As mentioned above, stimulating markets for digestate is very important and requires the maximising of nutrient content and the minimising of the presence of toxic and unwanted materials (including heavy metals, pathogens and inert materials). Whilst the process is an important aspect, the quality of the feedstock probably has the biggest effect and so it is vital to maximise its quality. Both admissible waste types and separation processes are important here.

There are a number of possible feedstocks which can be used in anaerobic digestion. These include the following: source separated food waste, sludge, agro-industrial by-products, manure, slurries, and yard waste. This study is concerned with MSW. One of the main limits on the AD process is its inability to degrade lignin (a major component of wood). This is in contrast with the process of aerobic biodegradation (i.e. composting). It is an important consideration when locating AD plants and designing collection methods.

Anaerobic digestion is better suited to waste with a higher moisture content than composting which requires 'drier' wastes. The process of AD can occur between 60% and 99% moisture content (IEA Bioenergy 1997*)*. Therefore kitchen waste and other putrescible wastes, which by themselves may be too wet and lacking in structure for aerobic composting, provide an excellent feedstock for AD. Woody wastes contain a higher proportion of lignocellulosic materials and may be better suited to aerobic composting. Liquids are often added to the AD processes (either water or recycled effluent) to maintain a high moisture content.

Any non-biodegradable components of MSW which are fed into an anaerobic digester, will not be affected by the process and are simply taking up unnecessary space. To maximise the benefit (both environmental and economic) and minimise the cost it is therefore important to minimise their presence in the AD feedstock. Minimising the quantity of potentially toxic materials is also an important consideration for the quality of the end-product. As with composting, the methods of MSW collection have a bearing on both.

There are two main alternatives for waste separation, and the choice between them has an important bearing on AD feedstock quality.

Source separation is actively encouraged in a number of member states. It includes the separation of the putrescible organic fraction (biowaste). It is generally accepted that source separation provides the best quality feedstock for both AD and composting with both a maximum organic content and minimum contamination with heavy metals, glass and plastics. After digestion in a reliable process, this will result in the formation of a quality digestate and a high volume of biogas.

Centralised separation is the only route for obtaining a digestible fraction from residual waste. The techniques involved include mechanical processing, optical processing and hand-picking. The digestible fraction obtained tends to be more contaminated than source separated biowaste with inevitable consequences for the digestate's ultimate utilisation.[7] There is also the risk of larger non-separated components of the waste causing physical damage to treatment plants further downstream (by abrasion, blockages or tangling).

2.6.6 Characteristics and Quality of End Products and Potential Markets

Biogas

The production of biogas from controlled anaerobic digestion is one of the principal advantages of the process. Stabilisation in the AD process is linked to methane production, with a theoretical methane production of 0.348 m^3/kg of Chemical Oxygen Demand (at standard temperature and pressure) being achieved by complete stabilisation (IWM 1998). In general, AD produces 100-200 m^3/tonne of BMW processed and it has a typical composition of 55-70% methane, 30-45% carbon dioxide and 200-4,000 ppm hydrogen sulphide.

Biogas generation is very sensitive to feedstock, one plant found volumes to range from 80 to 120 m^3 per tonne depending on waste input. There are also other constituents in smaller concentrations including carbon monoxide, hydrogen, nitrogen and oxygen. As mentioned above, a larger proportion of inorganics and polluting substances in the process will lead to smaller amounts of a 'dirtier' biogas.

The constituents of biogas (other than carbon dioxide and methane) can be quite important in its end-use. They include:

Hydrogen sulphide is particularly important where the gas is to be used for heat and power generation. Due to its corrosive qualities, care must be taken in the design of plant and the choice of materials which will come into contact with the gas. Removal of hydrogen sulphide from the biogas can be achieved by using iron salts in scrubbing

[7] There is some evidence that where pulping is used as a pre-process sorting phase, liquid separation can lead to removal of some more hazardous elements.

the gas, or adding them to the digester. Hydrogen sulphide provides the largest risk to personnel.

Hydrogen is a very important constituent in the process as it is used by methanogens to reduce carbon dioxide to methane gas.

Carbon monoxide is another important intermediate in the process and can be used as an indicator of heavy metal and organic toxification.

Because biogas is used to generate energy (possibly in the form of electricity and heat), it is the emissions after combustion that are important in terms of measuring its environmental impact. The biogas produced under optimum conditions has an energy content of around 20-25 MJ/m^3. Electrical conversion efficiencies will vary according to combustion plant. Practical experience with small-scale combustion engines with a rated capacity of less than 200 kW indicate an electrical conversion efficiency of around 25%, larger plants (up to 17,000 kW) can have a higher conversion efficiency of around 36%. There is also the added possibility of heating water from the engine's exhaust which can increase the overall conversion efficiency to 65-85% (IEA Bioenergy 1997).

Estimates concerning the utilisation of electricity by the plant vary a great deal. In rural AD plants, approximately 20 % of the electricity produced in the process is required for the plant operation while urban plants may utilise 2/3rds of the electricity produced. Biogas can also be both burned in boilers to produce hot water and steam for industrial purposes, and used as an alternative fuel in light and heavy-duty vehicles.

Digestate

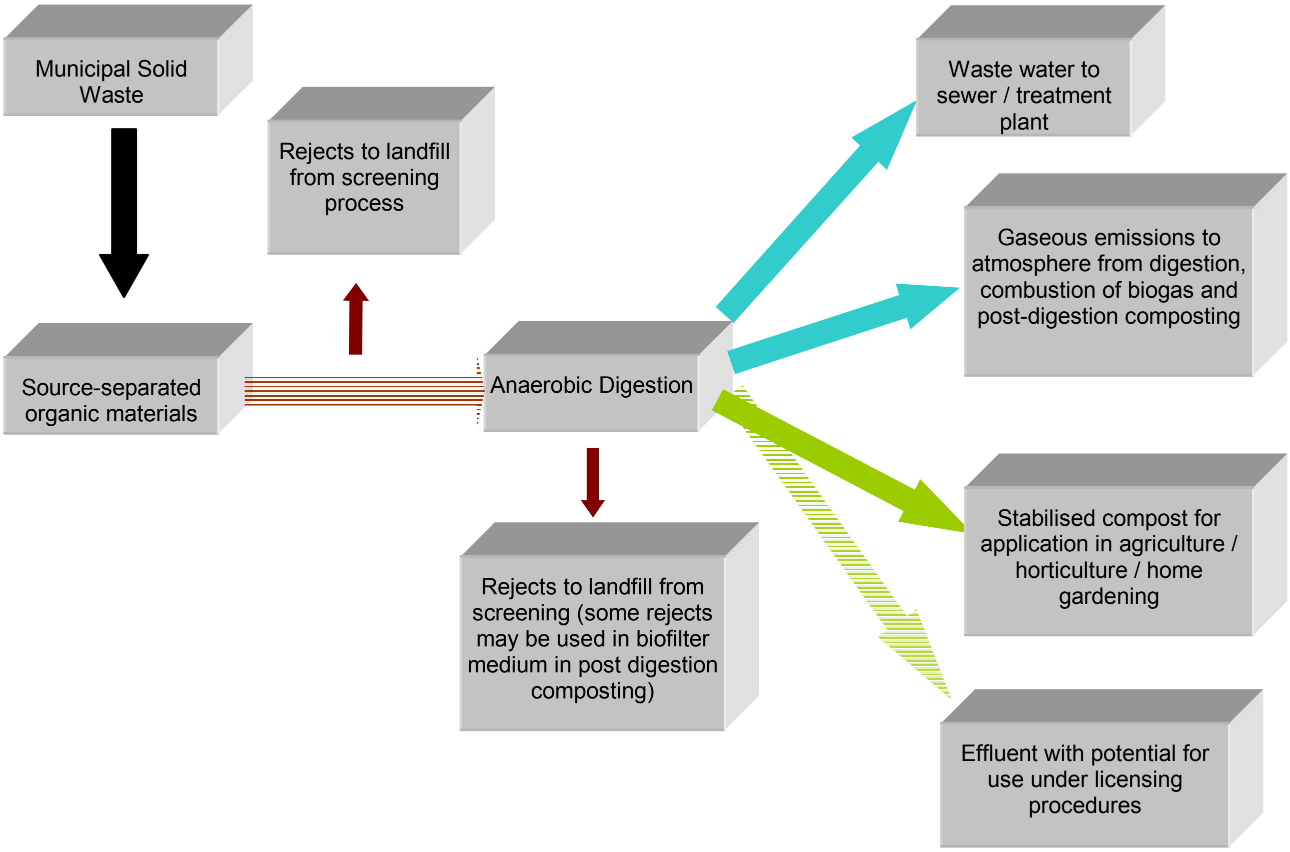
In order to extract the maximum recovery value from the organic waste input, the digestate from AD should have a useful purpose and positive benefit should be derived from its production. In practice this varies from use as landfill cover material, through direct application of the digestate to land for agricultural benefit, to further maturation and refinement of the digestate to produce a quality soil conditioner. The post-digestion composting phase is (as mentioned above) likely to be important to make the end product attractive to potential users in agriculture and elsewhere.

As well as the main product from the process, a solid digestate, small quantities of surplus liquor are also available which can be dewatered to provide liquid fertiliser. As stated above, the feedstock and process quality will have a significant effect on the characteristics of the end-products. A high nutrient content is important for use as a quality soil conditioner or fertiliser, as is a low concentration of potentially toxic materials such as heavy metals, inert materials and pathogens. Even so, the use of such materials may occur only under licence in some countries.

2.6.7 Northern and Southern Europe

As mentioned above, AD development in Southern Europe is likely to take place later than in Northern Europe (this is already evident) although the extent of diffusion in either case is difficult to gauge. Recent developments in source separation schemes in Italy and Spain paint an optimistic picture regarding the future availability of quality feedstock. It is also worth mentioning that AD is experiencing the fastest growth across Europe in Spain, thanks to public funding of facilities through EU funding programs, which reduce the overall management costs since depreciation is one of main cost items.

Figure 8: Schematic Representation of Anaerobic Digestion Inputs and Outputs



Note:
Red arrows represent residual materials
Blue Arrows represent 'negative outputs' (environmental costs)
Green Arrows represent 'positive' output (environmental benefits)

2.7 A Note on Paper Recycling

The 'recycling' of biowaste effectively implies composting / digesting the material, certainly as far as the terms of reference of this study are concerned. Paper and card, however, can also be recycled. These materials do fall under the scope of the Second Draft of the Working Document on the Biological Treatment of Biowaste.

The recycling of paper and card usually takes place with a specific end-product in mind. For this reason, waste paper and card tends to be categorised into different grades which are more and less appropriate for specific mills seeking to produce well specified end products. Waste paper and card is, however, not always used to manufacture paper. Other possible applications include:

- road surfacing (using cellulose fibres to help retain the bitumen and bind the stone-to-stone matrix);
- loose-fill heat insulation materials;
- moulded packaging (e.g. fruit trays, egg boxes);
- board products (e.g. noise insulation);
- animal bedding;
- automotive brake linings;
- refractory cement binders;
- artificial snow; and
- feedstock for ethanol production.

Many of these applications, whilst being relatively low value ones, require less sorting of paper and hence a cheaper feedstock. Recycling of paper and card into new paper and card products benefits from source separation and the higher quality of materials this provides.

All of these processes make use of energy, and generate emissions of various pollutants. Furthermore, they generate residues which, in the case of pulp, can be used in mills to generate energy. The question that is usually asked is whether the materials are more or less polluting than those processes with which they compete, or which they displace / replace.

Recycling is generally regarded favourably in terms of the economic analyses carried out thus far. However, this analysis has to be contextualised by consideration of the different materials, for which the analysis is quite distinct. There is little debate, generally, that the recycling of glass, aluminium and steel (less work has been done on textiles, which are often collected in recycling schemes) is positive, but the case for recycling of plastics and paper has been disputed on grounds that it may be better to combust these materials and generate energy from them rather than recycling them. In what follows, the emphasis is on the debate as it affects paper.

Regarding paper, the recycling of newsprint and magazines into secondary newsprint has generally received a favourable review on environmental grounds from researchers. This is the message of a number of studies carried out in this regard.[8] A review by Ecologika (1998) of the arguments put forward by those who propose combustion of waste paper portrays the argument not so much one of recycling versus incineration, but one between recycling, ending with incineration after the full economic life of the fibre, and 'premature incineration' (i.e. incineration prior to recycling). This is because modern secondary newsprint plants can make use of old fibre (which is too short to include in paper manufacturing) to generate their own energy.

[8] This includes work carried out by Broome et al (2000), ECOTEC and CSERGE (1999), Coopers and Lybrand et al (1997), Brisson and Powell (1995), Powell et al (1995) and Powell et al (1996).

It is also true to say that (as will be made clear) the time dimension of what actually occurs is critical in the overall environmental analysis, especially when considered in economic terms (because of the effects of discounting). To argue that carbon dioxide from incinerated paper can be absorbed by forests is to ignore the fact that trees take time to grow and to sequester the carbon released in the combustion process.[9] This is one reason why this study seeks to incorporate recognition of time as an important element in the economic analysis. Reporting all life cycle emissions as though they occurred simultaneously is misleading from the economic perspective. On the other hand, life-cycle analyses can illustrate a broader range of emissions than can currently be dealt with through economic analysis.

A recent study of the Aylesford recycled newsprint mill by Ecobilan (1998) comparing this with incineration of waste newsprint and magazines and the manufacture of primary newsprint (outside the UK) arrived at the following key conclusions:

- The Aylesford system required only 64% the primary energy used by the ‘incinerator system’;
- It generated 11% less carbon dioxide, was responsible for 44% less acidification and used 12% fewer hydrocarbons;
- The total contribution to global warming was 17% less than for the incinerator system over 20 years and 15% less over 100 years;
- The Aylesford system performed better in respect of eutrophication;
- The Aylesford system used more natural gas than the incinerator system but the total use of non-renewable resources was less;
- Regarding emissions to water, these had less nitrogenous matter and suspended matter and lower chemical oxygen demand than the incinerator system, but produced more phosphates;
- It used less water than the incinerator system;
- Regarding transport, the recycling system used only 37% the oil consumed by the incinerator system in transport phases. Virgin paper production (certainly for the UK market) relies on long distance transport. The recycling system emitted less than half the oxides of nitrogen than the incinerator system.

The study’s conclusions appear to support those reached in a number of others. Collectively, though space and resources preclude a more detailed discussion (and there are complex issues involved in these debate), the evidence tends to support the Commission’s ordering of rank in terms of treatments of paper, at least as regards ‘recycling’ versus ‘premature incineration’. It is not clear whether the production of, for example, moulded paper pulp packaging (from the old, shortened fibres) might be more beneficial than incineration. Nor does analysis undertaken thus far allow us to pronounce upon the relative position of composting of paper or card with respect to recycling or ‘premature incineration’.

[9] A USEPA report looked at the impact, on greenhouse gas emissions, of paper recycling. This work incorporated a model of the US forest sector to understand the impact of recycling. This also concluded that the recycling of paper had beneficial impacts in terms of greenhouse gas emissions (USEPA 1998). See also Smith et al (2001).

2.8 Overview

Table 2 gives an overview of the technologies discussed in the above Sections.

Table 2. Overview of Technologies for the Treatment of Biodegradable Municipal Waste

Overview of technologies for Biodegradable Waste	Biological Method			Thermal Method		
	Mechanical Biological Treatment	**Compost**	**Anaerobic Digestion**	**Incineration**	**Pyrolysis**	**Gasification**
Waste acceptance	Residual waste	Principally, source separated biodegradable waste since matter and nutrients are to be recovered with minimal contamination – composting of residual waste or separated fractions thereof increasingly uncommon	Principally, source separated biodegradable waste since matter and nutrients are to be recovered with minimal contamination – composting of residual waste or separated fractions thereof increasingly uncommon.	Residual waste	Most suitable for well defined dry waste fractions (from residual waste)	Source separated dry waste only unless combined with better cleaning technology
Acceptance of Wet Organic Fraction (kitchen wastes)	Technically, yes, but not applied to source-separated fractions	Yes, frequently conditional on presence of some structural material	Yes	Technically, yes, but not applied to source-separated fractions (especially given low calorific value)	Technically possible, but not likely to be applied to source-separated fractions	Technically possible, but not likely to be applied to source-separated fractions
Acceptance of Garden and Park Waste	Technically, yes, but not applied to source-separated fractions	Yes	Not usually	Technically, yes, but not applied to source-separated fractions (especially given low calorific value)	Yes	Possible
Acceptance of Organic Waste from Hotels and Restaurants	Technically, yes, but not applied to source-separated fractions	Yes	Yes	Technically, yes, but not applied to source-separated fractions (especially given low calorific value)	Yes	Possible but normally no

Overview of technologies for Biodegradable Waste	**Biological Method**			**Thermal Method**		
	Mechanical Biological Treatment	**Compost**	**Anaerobic Digestion**	**Incineration**	**Pyrolysis**	**Gasification**
Acceptance of Paper and Board	Technically, yes, but not applied to source-separated fractions	Yes	No	Technically, yes, but not usually applied to source-separated fractions which are readily recyclable	Yes	Possible
Excluded waste fractions	None	Metal, plastic, glass, mixed municipal waste as far as possible	Metal, plastic, glass, animal waste undesirable at plants without hygienisation, degradation of lignin requires post-digestion composting	None	Wet household waste	Wet household waste
Proven technology, track record	Yes; Very common	Yes; Very common	Yes; becoming common in some Member States	Yes; very common	Relatively few plants with long periods of continuous operation	Relatively few plants with long periods of continuous operation
Basic principle	Degradation by aerobic (and/or) anaerobic micro-organisms	Degradation by aerobic micro-organisms	Degradation by anaerobic micro-organisms	Combustion	Anaerobic thermo-chemical conversion	Thermo- chemical conversion
Cost of treatment	Costs of whole treatment depends upon destination of separated / treated fractions	Low to medium	Medium to high	Medium to very high	Medium to high	Medium to very high
Nutrient recovery	Yes 2.5 –10 kg N/ tonne of biowaste recovered 0.5 –1 kg P / tonne of biowaste recovered 1 – 2 kg K / tonne of biowaste recovered	Yes; 2 –4 kg N / tonne 1 – 2 kg P/ tonne 1 – 2 kg K / tonne	Yes; 4.0-4.5 kg N pr tonnes 0.5-1 kg P pr tonnes 2.5-3 kg K pr tonnes	No	No	No

Overview of technologies for Biodegradable Waste	**Biological Method**			**Thermal Method**		
	Mechanical Biological Treatment	**Compost**	**Anaerobic Digestion**	**Incineration**	**Pyrolysis**	**Gasification**
Energy recovery	Likely (e.g. through dry sabilisation / separation processes to manufacture RDF) Depending upon configuration, RDF may be (typically) 0.2-0.5 tonnes with calorific value around 15-20MJ/kg (sometimes higher). In addition, in some configurations, digestion processes can recover energy from degradation of biodegradable fractions (can be >100kWh depending on composition)	No	Yes; 100-250 kWh (0.4-0.9 MJ) per tonne of waste electricity In addition, CHP plants may generate a similar quantity of heat	Yes; Approx: 500kWh (2MJ) per tonne waste if electricity only CHP plants may generate lower electrical output but total energy recovered increases approx threefold (approx. 6-7MJ/tonne)	Yes; 0.7-1 MJ / tonne Also, energy containing product (char)	Yes; Approx. 2MJ per tonne waste
Total solid residuals, depending on waste (tonnes/tonnes waste)	0.7-0.9[1]	0.4 – 0.6	0.3 – 0.6	0.17 - 0.3	0.2 - 0.4	0.17 - 0.3
Quality products for recycling (recovery, tonnes/tonne waste)	Metals (0.05)[1]	Compost (0.5)	Fibres (0.3)	-	Char (0.2-0.4)	0.17 - 0.28
Other residuals possible for reuse with restrictions (tonnes/tonne waste)	RDF (0.3-0.4)[1] Stabilised organic fraction (0.07-0.2)[1]	-	Fluids (0.6)	Metals (0.05) Bottom Ash (0.15 – 0.22)	Grit, Glass, Slag, Metals and Chemical bulk)	Clinker, Grit, Glass, Slag, Metals and Chemical bulk
Residuals for land filling or other waste treatment	heavy and light rejects (0.2-0.4)[1]	Overflow sieving (0.02 – 0.1)	Overflow sieving (0.02 – 0.1)	Fly ash etc. (0.02 – 0.04)	(Char) (0.02-0.3)	Ash (0.03)

Source: Adapted from European Environment Agency (2001)

[1] *These figures depend upon detailed system configuration*

3.0 CURRENT SITUATION REGARDING MUNICIPAL WASTE - ARISINGS, COMPOSITION AND TREATMENTS

This Chapter considers the way in which municipal waste is currently treated in Member States. This question has to be considered in the light of appreciation of the variation in understanding, across Member States, as to what constitutes ‘municipal waste’. Two approaches were used to try to derive an accurate picture of the way in which municipal waste, and more specifically, biodegradable municipal waste is currently treated:

1. A review of published data; and

2. A survey of data available at the Member State level.

The latter is outlined briefly below.

3.1 Survey and Appraisal of Member State Data

In this study, each of the team members was designated responsibility for gathering data covering a sub-set of the countries being studied. The assignment of countries was based, as far as possible, upon existing experience of the team partners.

For each country, the aim was to elicit data and information regarding:

- Definitions of municipal waste;
- the composition of municipal waste – the information sources were the same as those above;
- the way in which municipal waste is currently treated in the country concerned
- the way in which municipal waste is likely to be treated in future (through reference to, e.g., plans for meeting the requirements of the Landfill Directive, or other strategies);
- the status of source separation of biodegradable municipal wastes; and
- the relative roles of composting and anaerobic digestion in treating source-separated fractions.

This survey work included seeking information from Member State Ministries, seeking publications concerning the waste situation in that country, and trying to project forward on the basis of existing and planed legislation, as well as published strategies. During this project, representatives of Member States and Accession States were encouraged to comment on the data derived following a meeting held by the European Commission in Brussels. Furthermore, the work has benefited from being able to use information emerging from a study on waste management policies in Central and Eastern European countries, an edited version of which has recently been published (Speck and Markovic 2001).

3.2 Member State Definitions of 'Municipal Waste'

The definition of municipal waste exhibits some variation across countries. However, for most countries, where there is a formal definition, municipal waste includes household waste (of all different types, including bulky materials) as well as commercial waste collected by the local authorities / municipalities concerned, and wastes generated from maintenance of parks, and street cleaning activities.

It is interesting that the definition of municipal waste does not exist as such in many countries, even though this is the fraction to which key EU legislation applies. It raises interesting questions (given the variation in definition) as to what the Landfill Directive, and any subsequent legislation, should be applied to. The reference data in the Landfill Directive is 'harmonised' Eurostat data from 1995, yet by common consent, there is no harmonised definition of 'municipal waste'. This raises interesting questions as to how the targets under the Landfill Directive are to be enforced, especially if the term is simply applied to 'all waste collected by municipalities'. This could lead to a situation in which, in an aim to comply with legislation, the responsibilities of municipalities are altered either through legislation, or implicitly, through discouraging collection of certain fractions (perhaps leaving them to be collected by private sector operators).

A clearer definition of municipal waste is required for use in legislation. The definition should ensure that the wastes being referred to are not subject to changes through what are essentially changes in administrative responsibilities of municipalities.

3.3 Quantities of Municipal Waste Arisings

Municipal waste arisings from Member States ought to be relatively well known. Member States are required to generate such information as a requirement of the Waste Framework Directive. In addition, planning for waste, and tracing the effectiveness of certain interventions, is impossible without adequate data.

There are issues, however, concerning data accuracy and comparability which make easy comparisons close to impossible. With regard to accuracy, the key point is, obviously, that weights of waste should be assessed before treatment. Certainly, it is not the case in all countries that waste is accurately weighed prior to treatment. In addition to the Waste Framework Directive, economic instruments and legislation such as the Packaging Directive will have focused more attention on the quality of data, and the Landfill Directive should also concentrate greater attention on municipal waste data (since without such data, progress towards Article 5 targets cannot be judged).

The treatment of home composting deserves mention. This is clearly of relevance to this study, yet estimates of the extent of active engagement in home composting, let alone, the weight of material diverted through this option, are generally not very reliable. Whether this should be included in the definition of 'waste' at all is questionable (it is usually regarded as waste minimisation as the material is not being 'discarded') but it is important to understand the extent to which this activity is being used to treat material which might otherwise be collected as part of the municipal waste stream.[10] Some municipalities in Germany and Austria may be treating 60% or more of their biowastes through home or community based composting. In such circumstances, to omit measurement of this fraction would be to give a false impression of the composition and quantity of material which has the potential to enter the municipal waste stream.

Eurostat has undertaken work in this area, whilst the OECD also collects data (see Table 3 below). The European Environment Agency, and the European Topic Centre for Waste in particular, have been keen to point out limitations in data comparability (because of the absence of common protocols and definitions). Countries such as Denmark and the Netherlands are reported to have household waste generation figures of about 500 kg/capita/annum. This is twice that reported for Iceland and Finland and 50% higher than that reported for Austria and Norway. Similarly, wide variations can be found for municipal waste.

The European Topic Centre lists the following possible explanations for variations:

- different definitions of waste and differences in systems used for waste data collection;
- differences in waste policy;
- differences in economic structure and lifestyle; and
- real differences in waste quantities produced.

The last of these is not so much an explanation as an observation. Other factors that will affect results are the level of home composting (since this is usually excluded from statistics) and the way in which waste is collected (larger containers may lead to more waste being collected, and garden waste collections may lead to material entering the waste stream that might otherwise have been home composted (or burned). For municipal waste, the extent of trade (i.e. commercial, industrial and construction and demolition wastes) is obviously a crucial factor. Lastly, it is somewhat odd that household numbers have not been given more serious consideration (as opposed to population numbers). To the extent that waste is produced by household units, the fragmentation of families, and the increasing 'individualisation' of society (which as a social phenomenon, affects different European countries to varying degrees), has been left under-explored.

[10] Where municipalities have introduced kerbside collections for green waste, this has usually led to major increases in waste collected. It is not always easy to trace whether this is due to a reduction in waste delivered to civic amenity / bulky waste collection sites, or whether this reflects the collection of materials which would otherwise have been composted at home.

Table 3. Generation of Household and Municipal Waste in EEA Member Countries (kg per capita).

Country	Household waste	Municipal waste
Austria	344	654
Belgium Brussels	366	655
Belgium Flanders	479	492
Belgium Walloon 1995	367	460
Denmark	496	540
Finland 1994	171	413
France 1995	435	597
Germany 1993	.	536
Greece	.	344
Iceland	242	558
Ireland	368	.
Italy	.	455
Luxembourg	.	461
The Netherlands	482	562
Norway	293	630
Portugal 1995	.	353
Spain	.	390
Sweden 1994	.	364
United Kingdom	442	476

Source: Eurostat 1999 (data for 1996 or latest year available according to OECD and Eurostat surveys).

3.4 Composition

Understanding the composition of waste is not straightforward. Waste composition varies not only across countries, but also by region according to:

- Socioeconomic status;
- Consumption habits;
- Season;
- Whether or not households have gardens;
- Presence (or otherwise) of tourists,

and many other factors besides. Furthermore, the reported statistics will be dependent upon methodologies used. Where fractions are co-mingled, and wet weights are reported, then transfer of moisture from one fraction to another is likely to occur (increasing the apparent quantity of paper and reducing the biowaste fraction). If the same waste was collected in separate fractions, the possibilities for 'weight transfer' would be reduced. Neither method is 'right' or 'wrong', but the different methodologies will be more or less relevant to different collection systems. Data are only comparable where the same methodology is used.

Presumably, as trends towards greater source separation continue, composition analyses should reflect that, becoming ever finer in their resolution. Furthermore, the further one moves up the waste management hierarchy, the more necessary it becomes to have statistical data that analyses composition at the level of products rather than broad types of material. At present, the categories used vary across countries.

Another factor affecting composition as reported below is what it is that is being analysed. Many compositional analyses are restricted to wastes collected at kerbside. However, as discussed in the previous section, municipal waste includes other wastes besides those collected at kerbside, so composition analyses may not reflect the totality of municipal waste (they are most often composition analyses of household waste as opposed to municipal waste). This is reflected in the classifications of material used in composition analyses, which seem inappropriate for the classification of bulky wastes. Also, different municipalities in different countries will collect different proportions of household waste at the kerbside. Some collect yard waste, others do not. Some encourage home composting, others do not. Hence, the proportion of the totality of materials potentially arising as waste which the analyses address will vary across countries, with corresponding effects on the composition stated.

Compositional data has been published by the OECD in its compendium. The results of this are shown in Table 4 below. Sweden and the UK were reported as having rather low biodegradable waste components, with France and Austria also below 30%.

Table 4. Waste Composition In EU Member States

Member State	Paper	Textiles	Plastics	Glass	Metals	Biodegradable waste		Others
Austria	670	63	340	284	166	750	29%	236
Belgium								
Denmark	505		122	94	42	923	36%	894
Finland	536			116	53	662	32%	735
France	6250	750	2750	3250	1000	7250	29%	3750
Germany								
Greece	640	144	272	144	160	1568	49%	272
Ireland								
Italy	3300		1050	900	450	6450	43%	2850
Luxembourg	36	4	15	13	5	83	44%	33
Netherlands	1785	230	395	445	230	2630	38%	1220
Portugal	805		420	175	140	1225	35%	805
Spain	3025	689	1511	984	589	6303	44%	1195
Sweden	1408	64	224	256	64	960	25%	224
UK	7400	400	2000	1800	1400	3800	19%	3200

Source: OECD (1997).

The reviews of Barth (2000) and Amlinger (2000) of biodegradable fractions of municipal waste again shows the UK, France, Ireland and Austria to be the only countries with biodegradable components less than 30% (see Table 5). Barth (2000) suggests an EU average figure of 32%. This is important since it suggests that of all fractions of the waste stream, this is usually the largest one (so that treatment separate collection potentially removes large quantities of material from other treatment options).

Table 5. Size Of Organic Fraction In European Municipal Solid Waste MSW

Country	Organic part in MSW (Barth)	Organic part in MSW (Amlinger)
Austria	29 % (1991)	29% hhld (1995) 17% MSW (1998)
Belgium	48 % Flanders (1996), 45 % Wallonia (1991)	Flanders: 48% (1996) Wallonia: 45% (1991
Denmark	37 % (1994)	37% (1994)
Finland	35 % (1998)	35% (1993)
France	29 % (1993)	29%
Germany	32 % (1992)	32%
Greece	49 % (1987 – 1993)	49%
Ireland	29 % (1995)	29%
Italy	32 – 35 % (1999)	33%
Luxembourg	44 % (1994)	44%
Netherlands	46 % (1995)	38%
Portugal	35 % (1996)	44%
Spain	44 % (1996)	44%
Sweden	40 % (1996)	25%
UK	22 % (1997)	21%
EU average	**32 %**	

Sources: Barth (2000) Amlinger (2000).

Investigations for this study have produced the figures presented in Table 6 below. These would need further investigation to check their accuracy (and their consistency). However, some comments and notes on the specific country data follow:

- It is clear that for most of the countries, the biowaste component constitutes between 35-45% of municipal waste. The exceptions appear to be Germany, Austria, Belgium (which is Flanders data) the UK and Ireland.
- However, the German and Austrian data do not include the relatively large quantities of waste believed to be composted at home in these countries. For both countries, home composting is believed to account for 10-12% of materials that would otherwise have to be collected (separately or otherwise). Hence, the composition of MSW that is biowaste in these countries might be expected to be lower than in other countries as the material is being ‘diverted’ from collection. Home composting is also widely practised in Flanders;

- Flanders data includes only the source separated fraction. As such, assuming two-thirds of the waste is captured, the total biowaste fraction could be expected to be probably 50% greater (or of the order 33%);
- UK waste composition data have been analysed in more detail elsewhere (ECOTEC 2000; House of Commons 2001). The official statistics are not reliable. They are based on old data capturing bin wastes only (approximately two-thirds of total MSW) using a methodology that has been subject to some criticism. Several localised compositional studies carried out in the UK show the biowaste fraction to be in the range 35-45%, in line with the rest of Europe. For example, 18 compositional studies carried out in London in 1998 gave an average figure of 38%, with London tower blocks yielding biowaste fractions in excess of 30% (Ecologika 1998). The basis for the Irish compositional data was not established. However, for the UK, it is proposed that the London data is used in this study in the absence of more robust data from official sources; [11]
- There is a suggestion that the figure for Ireland is as low as it s due to the fact that approximately one-third of the material to which the data refers is of commercial origin. This is reported in Ireland as having a 58.6% paper content. As such, the paper fraction is relatively high, and the biowaste fractions are small (the organics fraction in commercial waste is estimated at 15.1% as opposed to 32.9% in the household fraction);[12] and
- Particularly high figures for the paper and board fraction come from Ireland, the United Kingdom and Finland. The UK and Irish figures were discussed briefly above. One can speculate that in the UK, the explanation may simply arise from the fact that mixing of the waste leads to transfer of moisture from the biowaste fraction to the paper fraction. Arguably, this is not important where source-separation schemes are not in operation. The UK still operates with low levels of source separation, especially for biowaste fractions, and this may explain the lack of attention given to these issues in the past. Why the paper fraction should be so high in Finland is not known, though clearly Finland is a major producer of paper and paper products.

For comparison, Table 7 shows data recently gathered by the European Topic Centre for Waste. The Table shows the tonnages of biodegradable municipal waste as well

[11] The Environment Agency for England and Wales is to carry out a major research project into waste composition in the near future.

[12] The transparency of the Irish data leads one to speculate as to whether, given the fact that most municipalities are collecting trade waste, the compositional data are not generally lacking in their accounting for this fraction of municipal waste, which may be considerable as in Ireland.

as the fraction of municipal waste that is biodegradable. As in the data obtained for this study, the biodegradable fraction of MSW in Finland is very high reflecting the large paper fraction. The French figure is low because the data has been calculated using the figure for all municipal waste (excluding sewage sludge), whilst only bagged waste is effectively included in the biodegradable fraction. Were cleaning residues and trade wastes to be taken from the municipal waste figure, the biodegradable fraction would fall into line with that of most other countries.

Table 6. Quantity And Composition Of Municipal Waste In Member States And Applicant Countries (composition in % MSW)

	AU	BE	DK	FIN	FRA	GER	GRE	IRL	ITA	LUX	NL	POR	SPA	SWE	UK	CZ	CYP	EST	HUN	POL	SLO
YEAR (quantities)	1998	1997	1998	1997	1998	1998	1997	1998	1998	1997	1998	1999	1997	1998	1998/ 99	1998	1993	1996	1998	1996	1995
Total MSW (million tonnes)	4.85	4.69	2.93	2.51	44.4 (38)	49.1	3.9	2.06	26.9	0.30	8.22	3.8	17.2	3.81	31.5	3.24	0.37	0.56	4.3	11.8	1.02
Total (OECD) (1997 data, or latest year)	4.1	4.85	2.95	2.1	28.8	40.0	3.9	2.03	26.6	0.19	8.72	3.8	15.3	3.2	28.0	3.2 (2.6)			5.0	12.2	
Kitchen and Yard Waste	29.2	34.9	37	40	29	29.9	47	27	33.6	43.8	42.0	37	44.1	40	38.1	-	35	53.0	37.5	31.7	32.3
Yard Waste		*13.4*							*5.0*								*11*				
Kitchen Waste		*21.6*							*28.5*								*24*				
Paper and card	24	18.9		36.8		16	20	32.5	22.8	19.2	34.4	26	22.2	37	26.1	-	25	8.1	16.8	18.6	14.9
Paper																	*17*				
Cardboard																	*8*				
Timber	1.4	1.9 [1]											1	1.3		-					
Textiles	2.8	2.6		0.8		2		2.1	5.1	2.3	2.0	3	4.8	1	3.3	-			3.9		
Nappies						2.8			2.0	4.0						-					
Plastics	8.2	6.8		4.5		5.4	4.5	11.4	10.3	7.9	5.8	10	10.6	7	8.3	-	13	3.0	5.2	3.7	9.7
Glass	9.4	5.1		2.3		9.2	4.5	4.8	7.2	6.7	10.0	6	6.9	2.6	8.7	-	3	7.4	3.8	7.5	5.3
Metals	7.2	3.7		3.2		3.2	4.5	2.6	3.0	2.7	2.6	2	4.1	3.5	3.4	-	4	4.3	3.6	3.5	6.6
Ferrous metals								*1.7*			*2.1*		*3.4*		*2.9*						
Non-ferrous metals								*0.9*			*0.5*		*0.7*		*0.5*						
Other	17.8	26.0		12.4	37.3	31.5	15.5	19.2	15.9	12.9	4.2	16	6.6	6.6	12.1	-	20	24.2	29.2	36	31.4
YEAR (Composition)		1998		Early 90s		1993	1997	1998	1998	1994	1996		1997		Early '90s		1993	1996	1998		1995

Notes and Data Sources:

Austria:	*Umweltbundesamt, Klagenfurt 1998.* Compositional data applies to household waste, estimated at 2.8 million ones in 1996. The data excludes material which is home composted, estimated at around 0.3 million tonnes.
Belgium	Personal communication with OVAM staff. Compositional data is from Flanders only (which produces the majority of Belgium's waste)
Denmark	*Waste Statistics 1997, Environmental Review from the Danish Environmental Protection Agency, No. xx.* In addition to this household waste, some 0.86 million tonnes of waste from institutions / offices were collected. Danish EPA comment – no figures describing the total composition of MSW. They are planning a project to describe the total composition of MSW and to evaluate the present home composting solutions that some regions have chosen in order to reduce the amount of waste produced. The estimated fraction of kitchen and yard waste comes from Amlinger (2000).
Finland	*The National Waste Plan Until 2005*, Finnish Ministry of the Environment, 1999; Data from the Finnish Environment Institute. Composition data is from the early 1990s.
France	Data from ADEME. The French definition of municipal waste includes wastes from waste water treatment and sewage sludge. These account for an estimated 9.5 million tonnes in 1995. Hence, MSW was estimated at 38 million tonnes in 1995 (Amlinger (2000)). The 29% figure for Kitchen and Yard waste applies to 21 million tonnes of household waste.
Germany	*Prognos, 1999.* Compositional data excludes home composted materials, estimated at 4.5 million tonnes.
Greece	From the *National Plan for SWM* (data covering 30% of the country's population)
Ireland	*National Waste Database 1998.* Composition data refers to collected waste only (i.e. 1.85 million tonnes of the 2.06 million tonnes arising).
Italy	*ANPA estimates for 1998; Federambiente survey (2000)*
Luxembourg	Hofmann, F-J. (1996) *Hausabfallanalyse 1992-1994im Grosherzogtum Luxemburg*, in proceedings of Luxembourgeois Symposium sur le Gestion de Dechets, Luxembourg 1996. Composition refers to household waste only (circa 0.141 million tonnes)
Netherlands	*Ministry of Housing, Spatial Planning and the Environment, household waste only. Composition data are derived from author's calculation from composition of residual, and capture rates of separately collected materials.*
Portugal	Compositional data are from LIPOR for Lisbon.
Spain	*Medio Ambiente en España 1998*
Sweden	The two sets of compositional data are, firstly, from RVF in 1998 (samples of household waste excluding bulky materials) and, secondly, from RVF (1993) for MSW.
UK	Data from DETR (2000) *Municipal Waste Surveys*, SEPA (1999) Waste Strategy for Scotland, and Northern Ireland Heritage Service. Compositional data are based on Ecologika (1998)
Czech Rep.	Czech Ministry of the Environment
Cyprus	*World Bank: Republic of Cyprus. Environmental Review and Action Plan, 1993*; Evaluation report of present Waste Management situation in Cyprus, 1993, prepared by Carl Bro Environment a/s and NV Consultants
Estonia	*EST-101:Final Report* Estonia composition is waste landfilled only
Hungary	Hungarian Ministry of Environment
Poland	Polish Ministry for Environment
Slovenia	*Statistical Office of the Republic of Slovenia*

We are also grateful for having had access to information emerging from an ongoing project funded by the Regional Environment Centre in Szentendre concerning waste management policies in Central and Eastern Europe.

In Germany, the low figure for the biodegradable fraction emerges because the biodegradable fraction of municipal waste estimated here refers to the biodegradable fraction of household waste only. The figures would seem to indicate that a very large fraction of municipal waste in Germany arises from non-household sources. The low Italian figure is something of an anomaly since this was estimated on the assumption that only 35% of bagged municipal waste (i.e. residual waste) was biodegradable. In fact, the kitchen and yard waste fraction alone probably accounts for 35%, but the total biodegradable fraction (including paper, wood, biodegradable textiles etc.) most likely lies between 60-65% of the bagged waste (given that a relatively small, though increasing, fraction is separated at source). Note, lastly, that the figure used for municipal waste in Austria appears to be the household fraction only, whilst that for the United Kingdom is apparently for England and Wales only.

Table 7. Baseline Data For Landfill Directive

Country	Year	MSW	Biodegradable MSW	Landfilled Biodegradable MSW	Biodegradable Fraction of MSW
Austria	1995	2644	1495	302	57%
Belgium (Flanders)	1995	2890	1671	623	58%
Denmark	1995	2787	1813	205	65%
Finland	1994	2100	1664	928	79%
France	1995	36200	15746	5988	43%
Germany	1993	43486	12000	N/A	28%
Greece	1997	3900	2613	N/A	67%
Ireland	1995	1503	990	903	66%
Italy	1996	25960	9170	6821	35%
Luxembourg	1995	N/A	N/A	N/A	No data
Norway	1995	2722	1572	1069	58%
Portugal	1995	3340	N/A	N/A	No data
Spain	1996	17175	12196	N/A	71%
Sweden	1998	4000	N/A	N/A	No data
The Netherlands	1995	7105	4830	1365	68%
U.Kingdom	1996/97	25980	16366	14675	63%

Source: European Topic Centre for Waste (2001)

On the basis of this examination of the data, investigations for this piece of work appear to have generated a consistent dataset. Some reservations remain concerning the German, Austrian and French datasets in particular since in each of these cases, a large fraction of non-household waste appears to be collected. This will imply that the use of compositional data, most probably based on household waste, will not necessarily reflect the composition of municipal waste as a whole. For Austria, Amlinger (2000) suggests that the amount of organic wastes in total municipal waste is little different to the amount in household waste (despite a marked difference in total municipal and total household waste). Barth's (2000) figures for Germany indicate the same may be true for Germany (see below).

3.5 Relative Significance of Treatment Technologies

Data on treatment methods ought to be fairly reliable for the same reasons as discussed under waste arisings. As discussed in the section on composition, issues may arise in interpreting data on home composting and re-use. No country explicitly reported re-use, though deposit-refund schemes are in operation in several countries. Presumably, these are captured mainly in the recycling treatment approach (or in the relative absence of certain materials such as glass in the waste stream).

Table 8 shows the data collected through the review carried out for this project. The following comments can be made:

- There is considerable variation in treatment across countries. Some countries have already made significant moves towards materials recovery, others have not. Those that have made significant progress are Austria, Belgium (especially Flanders and Wallonia), Denmark, Finland, Germany, Netherlands, and Sweden;
- Reflecting this, a number of countries have already begun to compost significant fractions of their municipal waste, though progress here has not been as rapid as with dry recyclables (despite the fact that the relative proportions of dry recyclables and biowastes are similar - note that the figures here do not generally consider the impact of home composting, which as discussed earlier is subject to considerable uncertainty in its estimation); and
- Accession States (and some others dependent upon low-cost landfill solutions) face a dual problem in not only diverting material from landfill, but also, improving management of existing landfill sites.

Table 8. Treatment Of Municipal Waste In Different Member States

	AU	BE	DK	FIN	FRA	GER	GRE	IRL	ITA	LUX	NL	POR	SPA	SWE	UK	CZ[a]	CY[e]	ES	HU	POL	SL
Landfill	32	38	12.4	61.1	59	41	91.31	91	79.2	24.9	15.2	95	73.3	27	82.4	61.0	99	99	90	98[e]	95.9
Sanitary	*28.5*						*45.38*						*56.7*						*27*		
Dumping							*45.93*						*16.6*						*63*		
Incineration	14.7	30	58	5.5	29	19			6.5	50.6	40.5		5.9	38.1		5.3			6		
With energy recovery		*22*			*22*										*7.6*	*5.2*					
Without energy recovery		*8*			*7*	*19*[b]									*0.1*	*0.1*					
Recycling	34.3	20	12.4	28.2	5.8	24[c]	7.88	8.8	3.7	10.8	25.4		2.7	26.5	7.7	10.1	1		4		2.1
Anaerobic digestion	0.1		0.1	1	0.2	1								1.9							
Composting (source separated)	15.3	12	17.2	3.7		13			5.6	13.7	18.9	5	0.5	6.2	1.8	18.5					0.4
Composting (mixed waste)							0.81	0.2					17.5								
Biological Mechanical Treatment	6.3					1			6.1												
Storage																					
Home composting					6																
Year	1999	1996	1997	1997	1997	1998	1997	1999	1999		1996	Early 90s	1997	1998	1998/ 1999	1997			1998		1995

[e] Estimate
[a] Data for household waste. Between 0.7-2.25 million tonnes of non-household municipal waste is collected, of which an estimated 350,000 is composted in windrows.
[c] The without energy recovery category includes all incineration not defined as recovery in Germany (i.e. calorific valueless than 11 MJ per kg)
[d] These figures include all incineration classified as recovery (>11MJ per kg).

Sources:

Austria *Umweltbundesamt, Klagenfurt 2001 (data for household waste only)*
Belgium Estimated from European Environment Agency data (1996 being latest 'consistent' year across the Belgian regions)
Denmark *Waste Statistics 1997, Environmental Review from the Danish Environmental Protection Agency, No. xx.*
Finland Data from the Finnish Environment Institute
France *ANRED* and *ITOM-ADEME.* Statistics apply to waste received by public waste disposal sites and collectively transported to the site. Volumes of HIW not collected by municipalities are not included.
Germany *Prognos, 1999*
Greece From the *National Plan for SWM* (data covering 30% of the country's population)
Ireland *National Waste Database 1998*
Italy *ANPA 1999.*
Luxembourg Figures are based on an assumption concerning dry recyclables based on figures in BECO Milieumanagement & Advies, Antwerp, May 1999
Netherlands *Ministry of Housing, Spatial Planning and the Environment, household waste only. Composition data are derived from author's calculation from composition of residual, and capture rates of separately collected materials.*
Portugal No data – taken from OECD. This data is clearly out of date.
Spain *Medio Ambiente en España 1998*
Sweden *Swedish Waste Management 1998.*
UK Combination of *DETR Municipal Waste Survey*, *Waste Strategy Scotland* and Northern Ireland Heritage Department.
Czech Republic *Czech Environment Institute*
Cyprus *World Bank: Republic of Cyprus. Environmental Review and Action Plan, 1993*; Evaluation report of present Waste Management situation in Cyprus, 1993, prepared by Carl Bro Environment a/s and NV Consultants
Estonia *EST-101:Final Report*

Hungary	Hungarian Ministry of Environment
Poland	See final note below
Slovenia	Statistical Office of the Republic of Slovenia

We are also grateful for having had access to information emerging from an ongoing project funded by the Regional Environment Centre in Szentendre concerning waste management policies in Central and Eastern Europe.

3.6 Importance of Separate Collection

At a general level, the rationale for source-separation of wastes is that it makes it easier to deal with, and to recover value from, the resulting streams in the most appropriate manner. The quality of the materials recovered tends to be higher because of lower levels of cross-contamination with other components of the waste stream. Arguably, determining the best option for specific components of the municipal waste stream implies that they are collected separately. Furthermore, there is anecdotal evidence to support the view that the act of separation increases awareness amongst citizens of waste as an environmental issue. It is, after all, impossible to separate waste into specific fractions without first assessing what waste one has to separate in the first place. The act of separation thus becomes an activity which serves to educate citizens as to what wastes they produce, and in what quantities.

Experiences with attempts to derive ‘compost’ from mechanical separation of mixed residual waste have tended to reveal two things:

- The end-product is generally of a low quality, and has led to problems (and rejection) when attempts have been made to use the material on land. Typical contaminants are heavy metals, glass and plastics;
- The attempt to market such materials can severely compromise the market for materials of higher quality, derived from source-separated materials. It is no co-incidence that in countries where the utilisation of compost from source separated municipal wastes is furthest advanced, the attempt to derive ‘usable products’ from residual wastes is frowned upon (or banned, other than for quite specific, low-grade applications) for exactly this reason.

The situation in respect of separate collection of biodegradable waste is, therefore, reflected in the development of standards for compost and composting processes. In this respect, one can say that systems of standards and the systems for separate collection co-evolve. Standards facilitate the marketing of compost since they give potential users confidence in the quality of the materials they are buying, ensuring fitness for purpose of the composted products concerned. Equally, separate collection is seen as a pre-requisite for producing composts of a consistent and reliable quality.

The biodegradable fraction of waste (as defined here) is of particular significance in the context of waste management. The cycling of carbon, which in ‘wilderness areas’, occurs naturally and in the course of time, has been seriously altered by the activities of human beings. In particular, the disturbance of soils leads to the oxidation of soil carbon with the release of carbon dioxide into the atmosphere. The quantity of carbon effectively emitted to the atmosphere over time (relative to this ‘wilderness’ baseline) due to soil disturbance is believed to be enormous.

The consumption of increasing quantities of crops grown in this way has continued to grow (with population growth and the demand for protein-rich diets) yet in most (perhaps all) countries, organic matter is not being returned to soils at the same rate at which it is being removed (and removal occurs both in the form of the crops themselves, and the oxidation processes mentioned above, which are related to tillage of the soil). Farm mechanisation and consequent specialisation has also facilitated regional specialisation in arable and livestock farming units such that just as soils are being depleted of organic matter in some areas, excess production of organic sources of nutrients occur in others. Estimates show that topsoil C levels are 20-50 percent of their pre-cultivation values in mineral soils converted to agriculture. These losses are much higher for organic soils converted to agricultural use. If some or all of this carbon were reabsorbed by agricultural soils, this would constitute significant mitigation.

Quite apart from the carbon content of materials, biodegradable materials contain nutrients which are also of value to the soil. Indeed, the fact that carbon has been lost in soils over time can be traced partly to the increasing dependence of agriculture on inorganic fertilisers, synthesised specifically for the purpose of soil fertilisation. The primacy of the production of synthetic fertilisers for agricultural production reflects the evolution of disciplines in the field of agricultural science.

There are believed to be a number of potential advantages associated with the use of organic matter in the soil, and those related to the biological properties of / associations with organic material are likely to be somewhat less well-accepted / appreciated for the reasons mentioned above. Benefits of using organic materials are discussed further elsewhere in this report, but they include:

- Improved soil structure, porosity and density, improving root environment;
- Increased infiltration and permeability, reducing runoff and erosion;
- Improved water holding capacity, reducing water loss and leaching in sandy soils;
- Supply of macro and micronutrients;
- Control / suppression of soil-borne pathogens;
- Organic matter;
- Cation exchange capacity of soils / growing media improved (so increasing ability to hold nutrients for plant use);
- Supply of beneficial micro-organisms to soils and growing media;
- Improves / stabilises soil pH;
- Potential to bind and degrade some pollutants; and
- Potential to facilitate associations with mycorrhyzal fungi in soil (which are important in facilitating the uptake of micronutrients from the soil.

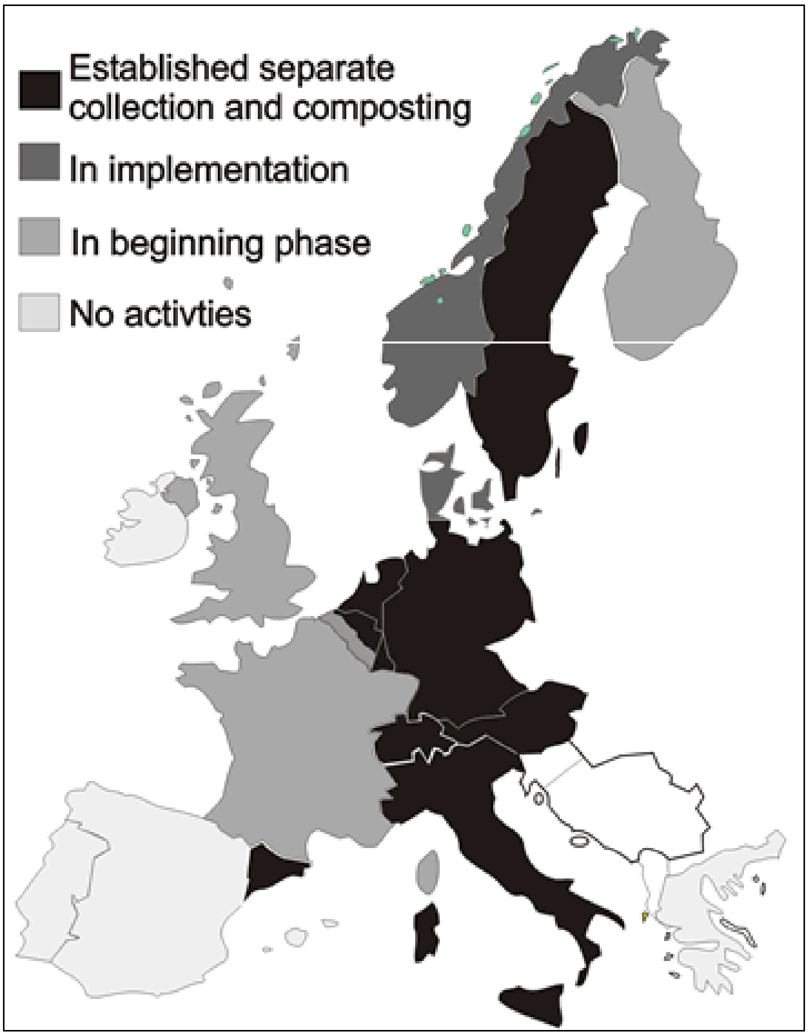
These suggest there may be considerable benefits from returning organic matter to soils in the form of composted materials. These are examined in further detail in Appendix 4.

3.7 Status of Separate Collection in Member States and Accession Countries

Some Member States and Accession Countries are already well advanced in terms of putting in place systems for the separate collection of biodegradable municipal waste (see Figure 11 below). These Member States tend also to have in place a system of standards for treatment processes and / or end products for compost and digestate.

At the other extreme are Member States and Accession Countries who are currently doing very little in the way of composting / digestion. These countries tend also to have little in place by way of standards for processes / end-products.

Figure 11: Stage of Implementation of Source Separation for Organic Waste in Europe



Source: Barth (2001).

Several studies have been undertaken in the past by DHV et al (1997), Barth (2000), FEAD (for International Solid Waste Association, u.d.), Amlinger (2000) and the European Topic Centre for Waste (2001), whilst some data is also published in the Vienna Workshop report (Federal Ministry for the Environment, Youth and Family

Affairs 1999).[13] In Table 9, the results of investigations carried out for this project are shown alongside the results of other work.

It is interesting to compare collected quantities with the potential for collection (on the basis of the putrescible content of municipal waste as outlined in the discussion above – see Table 10). This serves as a cross check on the municipal waste data and composition data. It shows the countries with the best rates of collection to be Denmark, Austria, Belgium (especially Flanders – see footnote in Table), Germany, Luxembourg and the Netherlands with the Czech Republic also performing well. This does not show the degree to which home composting plays a role. This is clearly considerable in Austria, Germany, Flanders, Luxembourg and possibly for some other countries too. Swedish estimates are for 40,000 tonnes of home composting.

[13] We are grateful to the ISWA Working Group on Biological Treatment for allowing us to access the FEAD data.

Table 9. Status Of Separate Collection Of Compostable Fractions ('million tonnes)

		This Study			Barth			DHV		Amlinger (2000)		FEAD (2000)		
		Amount		**Year**	**Amount**		**Year**	**Amount**	**Year**	**Amount**	**Year**	**Amount**		
		Doorstep	**Green only**		**Doorstep**	**Green only**				**Bio/Green waste**		**Doorstep**	**Green only**	**Mixed**
Austria		0.36[b]		1996	0.88[a]	0.85[a]	1996	0.88[a]	1996[a]	**0.60**	1998/99	-		-
Belgium												0.265	0.282	0.95
	Wallonia	0.088		1997	0.12		1994	0.163	2002 (?)	0.120	1996			
	Flanders	**0.344** **0.752**	**0.388**	**1998** **1999**	0.33	0.39	1998	0.09	1995	0.738	1999			-
	Brussels			1996										
Denmark		**0.051**[c]	**0.586**[c]	**1998**	0.028	0.49	1997	0.034	1995	0.520	1997	0.035	0.434	-
Finland		**0.093**	**-**	**1997**	0.1		1998	0.069	1995	0.100	1998	-		-
France		**0.07**	**0.9**		0.08	0.76	1998	-	-	0.861	1998	0.04	0.5	1.7
Germany		5		2000	**7**		**1998**	**-**	**-**	**7.000**	1998	6.4		-
Greece		**0.03**	**Mixed waste only**	**1999**							1995			
Ireland		**0.006**[c]		**1998**	-		-	-	-		1998	0		-
Italy		**1.5**	**-**	**1999**	1.5		1999	-	-	1.500	1998/99	0.405		-
Luxembourg		**0.034**	**-**	**1998**	0.03		1998	0.007	1994	0.030	1998	0.025	0.05	-
Netherlands		**1.5**	**0.29**[c]	**1998**	1.5	0.8	1996	1.45	1995	1.800	1998	1.5	0.4	-
Portugal			**0.014**[c]	**1998**										
Spain		0.013 (Cat)	-	1999	0.06 (Cat)		1998	-	-	**0.040 (Cat)**	**2000**	-		0.302
Sweden		0.119[c]	0.15	1998	0.13	0.15	1997	0.04	1994	**0.400**	**1999**	0.05	0.15	0.02
UK		**0.042**	**0.573**	**1998**	0.039	0.86	1998	-	-	0.910	1998	0.013	0.3	-
Cyprus		**None known**												
Czech Rep.		**0.01**	**0.375**	**1999**										
Estonia		**0.001**		**1998**										
Hungary		**None known**			-									
Poland		**0.222**		**1999**										
Slovenia		**0.01e**		**1999**										

Sources: European Commission DG XI (1997), Barth (2000), Amlinger (2000), European Topic Centre for Waste (2001) Federation Europeenne des Activites du Dechet et de l'Environment (u.d.)
Notes: The figures emboldened are believed taken to be the best estimates
[a] The Austrian data includes home composting.
[b] The information obtained by the study team is consistent with that from Barth, DHV et al and Amlinger in that it appears to be for separate collection from the doorstep only. DHV note that 0.36Mt is collected from biobins while Amlinger gives a similar figure.
[c] These figures are taken from the report European Topic Centre for Waste (2001)

Table 10. Proportion Of Available Organic Municipal Waste Materials Treated Through Separate Collection

	AU	BE	DK	FIN	FRA	GER	GRE	IRL	ITA	LUX	NL	POR	SPA	SWE	UK	CZ	CYP	EST	HUN	POL	SLO
Total MSW (million tonnes)	2.800	4.690	2.780	2.510	28.000	49.100	3.900	2.060	26.850	0.25	8.220	3.800	17.200	3.810	31.500	3.240	0.370	0.560	4.300	11.800	1.020
Kitchen and Yard Waste (%)	0.292	0.350	0.350	0.400	0.350	0.299	0.470	0.270	0.336	0.438	0.420	0.370	0.441	0.400	0.381	0.400	0.350	0.530	0.375	0.350	0.323
Theoretical Collection Potential	0.650	1.310	0.780	0.800	7.840	11.740	1.470	0.440	7.220	0.09	2.760	1.120	6.070	1.220	9.600	1.040	0.100	0.240	1.290	3.300	0.260
Current Level of Collection	0.360	0.840	0.490	0.093	0.970	5.000	0.000	0.000	1.500	0.034	1.500			0.264	0.615	0.385		0.001		0.222	0.010
Current as % Maximum	55%	64%	63%	12%	12%	43%	0%	0%	21%	39%	54%	0%	0%	22%	6%	37%	0%	0%	0%	7%	4%

Notes:
Amlinger (2000) gives a figure of 68% for Austria (1998/99) based on figures of 4.86 million tonnes MSW with a composition 17% biodegradable.
Amlinger (2000) gives a figure of 82% for Flanders (1999)
Barth (2000) gives a figure of 77% for Germany (1998)
If ones uses official composition data for England and Wales, the figure rises to 9%.
The key discrepancies across datasets probably relate to the application of what are probably household waste compositions to the total municipal stream. For those countries where large quantities of non-household waste are collected in the municipal fraction, the errors which are incurred in making this assumption are likely to be considerable.

3.8 Standards for Composting in EU Member States and Accession Countries

The situation as regards standards for compost has been presented, and recently reviewed, by Amlinger (1999; 2000).[14] This has been updated, and added to on the basis of the review from Member States carried out for this study. The italicised data represents 'new / additional information' relative to Amlinger's work (see Tables 11-17). Those countries reporting no compost standards include:

- France these are in preparation
- Luxembourg currently uses the German RAL standards – standards in preparation
- Hungary : MoE Decree now being drafted will provide legal framework

3.8.1 The Importance of Standards

One of the most important aspects of the success of biological treatment processes as an integrated part of waste management, is demand for the end-product. There are many different uses for compost but common to all is the need for confidence in the quality of the product. In European Member States, the approach to setting compost standards varies in the extent to which these are made statutory.

If there is such a thing as an emerging tendency in the setting of compost standards, it is that:

- statutory or quasi-statutory standards are established, which cover characteristics deemed important for the protection of the environment and human health; and
- voluntary systems of quality assurance support these standards, whilst also seeking to move beyond them through making recommendations regarding fitness of products for different end uses (for example, by matching compost products with different agronomic characteristics to specific end-uses).

The statutory standards usually establish limit values for potentially toxic elements and other characteristics designed to ensure the material has been composted. The voluntary standards typically concentrate on agronomic features and on quality control of the compost plant. The distinction, however, between statutory and voluntary, and exactly where the one ends and the other begins, varies across countries. Furthermore, across OECD countries, the approach to the setting of statutory

[14] A further updated version of the standards in existence, including revisions incorporated in the Austrian Biowaste Ordinance, will be available shortly (Hogg et al 2002, forthcoming).

standards also varies with the US, for example, following an approach to setting limit values for toxic element based more upon risk assessment than precautionary approaches based upon the load of, e.g., heavy metals applied to the soil.

Table 11. Heavy Metal Limit Values for Compost and Sewage Sludge in European Countries, Canada, USA and New Zealand (all values in mg/kg d.m.)

Country	Regulation	Cd	Cr_{tot}	Cr^{VI}	Cu	Hg	Ni	Pb	Zn	As	Co
EC/'eco-label'	488/98 EEC	1	100		100	1	50	100	300	10	-
EC/'eco-agric'	2092/91 EEC	0.7	70	0	70	0.4	25	45	200		
EC/'sew.sludge'	Lower limit	20	-	-	1000	16	300	750	2500	-	-
	Upper limit	40	-		1750	25	400	1200	4000	-	-
Austria	ON S 2200 Class I[2)]	0.7	70	-	70	0.7	42	70	210	-	-
	ON S 2200 Class II[2)]	1	70		100	1	60	150	400	-	-
	ON S 2200 Class III[2)]	4	150	-	400	4	100	500	1000	-	-
	Draft comp.ord.:agric	1	70	-	150	0.7	60	150	500	-	-
	Draft comp.ord.:reclam.	3	250	-	500	3	100	250	1200	-	-
Belgium	Min.f.Agric	1.5	70	-	90	1	20	120	300	-	-
Denmark	Sew.sludge	0.8	100	-	1000	0.8	30	120	4000	-	-
	Comp.before 01 06 2000	0.8	-	-	-	0.8	30	120	-	-	-
	Comp.after 01 06 2000	0.4	-	-	-	0.8	30	120	-	-	-
Finland	Trigger Values[o1)]	3	300	-	600	3	100	150	1500	50	-
	Target values 1998[o1)]	1.5				1		100			
	Fertilised growing media	0.5	-	-	100	0.2	60	60	150	10	
France	NF Comp.MSW	8	-	-	-	8	200	800	-	-	-
	Legisl.sew.sludge/industr.waste[oo]	20	1000	-	1000	10	200	800	3000		
	Standards currently being developed										
	From 2001	15									
	From 2004	10									
Germany	M 10 K.I[2)]	1.5	100	-	100	1	50	150	400	-	-
	M 10 K.II[2)]	2.5	200	-	200	2	100	250	750	-	-
	RAL GZ 25 I[2)]	1.5	100	-	100	1	50	150	400	-	-
	Blauer Engel[2)]	1	100	-	75	1	50	100	300	-	-
	Bio waste ordinance (I)[o]	1	70	-	70	0.7	35	100	300	-	-
	Bio waste ordinance (II)[o]	1.5	100	-	100	1	50	150	400	-	-
Greece [I), (x)]		*10*	*510*	*10*	*500*	*5*	*200*	*500*	*2000*	*15*	-
Hungary[(x)]		*2*	*100*	-	*100*	*1*	*50*	*100*	-	*10*	
Ireland	Sew.sludge;agr.use	20	-	-	1000	16	300	750	2500	-	-
	Compost limits[(x), (y)]	*1.5*	*100*	-	*100*	*1*	*50*	*150*	*350*	*15*	-
Italy	DPR 915/82	10	500	10	600	10	200	500	2500	10	-
Annex 748/84:	green compost	1.5	-	0.5	150	1.5	50	140	500		
	(peaty) composted amendment	1.5	-	0.5	150	1.5	50	140	500		
Luxembourg[1)4)]	Recommended	20	1000	-	1000	16	300	750	2500	-	-
	Limit value	40	1750	-	1750	25	400	1200	4000	-	-
	The Netherlands[3)]	1	50	-	60	0.3	20	100	200	15	-
	Compost										
	Compost (very dean)	0.7	50	-	25	0.2	10	65	75	5	-
Portugal[1)]		-	-	-	-	-	-	-	-	-	-
Slovenia		*Implementing Austrian compost regulations*									
Spain	Decr.1310/1990	40	1500	-	1750	25	400	1200	4000	-	
	pH>7[1)]										

Country	*Regulation*	*Cd*	*Cr_{tot}*	*Cr^{VI}*	*Cu*	*Hg*	*Ni*	*Pb*	*Zn*	*As*	*Co*
	pH>7[1)]	20	1000	-	1000	16	300	750	2500	-	
	B.O.E.n'm.131.2 June 1998	10	400	-	450	7	120	300	1100	-	
Sweden[1)]		2	100	-	600	2.5	100	100	800	-	
	Guideline values of QAS	1	100	-	100	1	50	100	300		
UK	Sewage sludge/pasture land	-	-	-	-	-	-	1.000	-	-	
	UKROFS fertil.org.farming	10	1000	-	400	2	100	250	1000	-	
	Composting Association Quality Label	*1.5*	*100*	-	*200*	*1*	*50*	*150*	*400*	-	
Canada[2)]		20	-	-	-	5.0	180	500	1850	75	0
USA[1)5)]	EPA/High Quality	39	(1200)	-	1500	17	420	300	2800	41	
	EPA/Others	85	3000	-	4300	57	420	840	7500	75	
	Composting Council 1996	39	-	-	1500	17	420	300	2800	41	
	Recommend. USDA (Min.f.agric.)	21	-	-	-	-	-	-	-	54	
New Zealand		15	1000	-	1000	10	200	600	2000	-	

1) Limit Values for sewage sludge in agricultural use
2) referring to 30% o.s.
3) >20% o.s. in d.m.
4) in preparation: values referring to RAL GZ 251
5) for all organic waste
° related to maximum application rate of 20 and 30 t.d.m./ha (3a respectively)
°° Cr+Cu+Ni+Zn: max. 4000 mg/kg d.m.o.

(x). taken from partner data

(y). also have limits for Mo = 5 mg/kg d.m., and Se = 2 mg/kg d.m.

Table 12. Limit Values for Heavy Metal Concentrations in the Soil Comparing the EU, Canada, USA and New Zealand (mg/kg d.m.)

Country	Regulation	Cd	Cr	Cu	Hg	Ni	Pb	Zn	As	Mo	Se
EC/sludge	Lower limit	1.0	100[7)]	50	1.0	30	50	150	-	-	-
	Upper limit	3.0	150	140	1.5	75	300	300			
Austria	ON L 1075	1.0	100	100	0.7	60	100	300	-	-	-
Belgium	Flanders[1)6)]	1.2	78	109	1.3	55	120	330	-	-	-
	Wollonia[1)]	1.0	100	50	1.0	50	100	200	-	-	-
Denmark		0.5	30	40	0.5	15	40	100	-	-	-
Finland[1)]		0.5	200	100	0.2	60	60	150	-	-	-
France[1)]		2	150	100	1.0	50	100	300	-	-	-
Germany[5)]	Clay	1.5	100	60	1	70	100	200	-	-	-
	Loam	1	60	40	0.5	50	70	150	-	-	-
	Sand	0.4	30	20	0.1	15	40	60	-	-	-
Greece		-	-	-	-	-	-	-	-	-	-
Ireland		1.0	-	50	1.0	30	50	150	-	-	-
Italy		3.0	[4)]	100	2.0	50	100	300	-	-	-
Italy[1)]		1.5	-	100	1.0	75	100	300	-	-	-
Luxembourg[1)]	Recommended	1.0	100	50	1.0	30	50	150	-	-	-
	Limit value	3.0	200	140	1.5	75	300	300	-	-	-
The Netherlands[1)]		0.8	100	36	0.3	35	85	140	29	-	-
Portugal		-	-	-	-	-	-	-	-	-	-
Sweden[1)]		0.4	30	40	0.3	30	40	75		-	-
Spain	1310/1990 pH>7[1)]	3.0	150	210	1.5	112	300	450	-	-	-
	pH>7[1)]	1.0	100	50	1.0	30	50	150	-	-	-
United Kingdom[1)4)]		3.0	400[3)]	135[2)]	1.0	75[2)]	300	300[2)]	50	4	3
	UKROFS	2.0	150	50	1.0	50	100	150	-	-	-
Canada		2.0	-	-	0.5	18	50	185	7.5	2.0	1.4
USA[8)]		19.5	1500	750	8.5	210	150	1400	20.5	9	50
New Zealand		3.0	600	140	1.0	35	300	300	-	-	-

1) Limit Values for application of sewage sludge
2) Soil pH 6.0 – 7.0
3) preliminary
4) in preparation: values referring Chromium (VI) 3mg/kg and chromium (III) 50mg/kg
5) Compost Ordinance
6) soil with 10% clay and 2% OM
7) planned
8) EPA 503 proposed regulation

Table 13. Admissible load of heavy metals in several European countries and the USA (kg/ha.y)

Country	Regulation	Cd	Cr_{tot}	Cr^{VI}	Cu	Hg	Ni	Pb	Zn	As	Mo
EC/'sew.sludge'*		0.15	3.0[4)]	-	12	0.1	3.0	15	30	-	-
Austria	Sewage sludge[1)]	0.02	1.25	-	1.25	0.02	0.25	1.0	5.0	-	-
	Fertiliser ordinance	0.001	0.63	-	0.63	0.001	0.038	0.63	2.5	-	-
	ON S 2200[2)]	0.015	1.5	-	1.5	0.015	0.9	1.5	4.5	-	-
	Draft com.,:agric[2)]	0.008	0.56	-	1.2	0.006	0.048	1.2	4.0	-	-
	Draft comp.:reclam.[2)]	0.02	1.67	-	2.67	0.02	0.67	1.67	13.3	-	-
Belgium	VLAREA	0.012	0.5	-	0.25	0.01	0.1	0.6	0.6	0.3	-
Finland	From 1995	0.003	0.3	-	0.6º	0.002	-	0.15	1.5º		
	From 1998	0.0015				0.001		0.1			
France**	s.sludge/industr.waste	0.03ºº	1.5		1.5	0.015	0.3	1.5	4.5		
	pH <6	0.015	1.2		1.2	0.012	0.3	0.9	3		
Germany	Sewage sludge	0.016	1.5	-	1.3	0.013	0.3	1.5	4.1	-	-
	Bio waste ordinance (I)[2)]	0.01	0.7	-	0.7	0.007	0.35	1.0	3.0		
	Bio waste ordinance (II)[2)]	0.01	0.67	-	0.67	0.007	0.33	1.0	2.67		
Italy		0.015	2.0	0.015	3.0	0.015	1.0	0.5	10	0.1	
The Netherlands[2)]	Compost	0.006	0.3	-	0.36	0.002	0.18	0.6	1.2	0.09	-
Spain	Decr. 877/1991ººº	0.15	3.0	-	50	0.1	3.0	15	30	-	-
Sweden	From 1995	0.0018	0.1	-	0.6	0.0025	0.05	0.1	0.8	-	-
	From 2000	0.0008	0.04	-	0.3	0.0015	0.025	0.025	0.6	-	-
United Kingdom		0.15	15[5)]	-	7.5	0.1	3.0	15	15	0.7	0.2
USA[6)]		1.9	150	-	75	0.85	21	15	140	2.0	0.9

* Directive 86/276/EEC; average within a period of 10 years

** max. mean load per year within a period of 10 years; Cr+Cu+Ni+Zn: 6kg/ha.y (Ph <6,0:4 kg/ha.y Se on pasture land: 0.12 kg/ha.y

1. sew.Sludge Ordinance, Lower Austria (Class III)
2. calculated from maximum compost dosage/ha*a
3. Maximum average within a period of 10 years
4. Planned
5. Preliminary
6. Only for sewage sludge that exceeds "high quality"
7. For secondary raw materials

º if deficit of Cu or Zn, the addition may be max doubled

ºº 0.015 kg/ha.y from 2001

ººº over a period of 10 years

Table 14. Provisions For The Exclusion Of Pathogens And Germinating Weeds And Plant Propagules In Several European Countries

	Indirect			Direct methods		
	°C	% H_2O	Days	Application area	Pathogens / weeds	Product (P) / approval of technology (AT)
EC/'eco-label'				Gardening	Salmonella sp.	None
488/98 EEC					E.coli	<1000 MPN (most probable number)/g
Austria draft	60	40	7	Land reclam.	Salmonella sp.	None
	55		14			
Comp.ordinance				Agriculture	Salmonella sp.	Non
					E.coli	If ded., recomm. For the safe use
				Sacked, sport / playground	Salmonella sp.	None
					E.coli	None
					Camylobacter,	None
					Yersinia sp., Listeria	None
				Technical use	-	No requirements
				Reclam. + agric.	Weeds/propagules	Germination ≤ 3 plants /l
Belgium VLACO	60	40	3		General	Non
					Eelworms	None
					Weeds	None
Denmark	55		14			
Finland	No harmful micro-organisms to such an extent that they may endanger man, animals or the environment					
France	60		4			
						(AT):[c)]
Germany Bio waste	55	40	14		Salmonella senft	None
ordinance	60[a)]	40	7		Plasmodoph. Brass.	Infection index: ≤ 0.5
					Nicotiana virus 1	Guide value bio-test: ≤ 2%
	65[b)]	40	7		Tomato seeds	Germination rate/sample: ≤ 2%
						(P):
					Salmonella senft	None in 50g sample
					Weeks/propagules	Germination ≤ 2 plants/l
Greece[(x)]		*40*			*Enterobacteriaceae*	*None*
Hungary[(x)]					*Faecal Coliforms*	*<10/g*
					Faecal Spectroccochi	*<10/g*
					Salmonella	*Not present in two 10g samples*
					Human Parasites	*Not present in 100g sample*
Ireland[(x), (q)]			*21*		*Salmonella*	*None (<3 MPN/4g)*
					Faecal coliforms	*< 1000 MPN/g*
					Plasmodiophora brassicae	*None*
					Tobacco-mosaic-virus (TMV)	
					Tomato seeds	*None*
						None
Italy	55		3			
Fertil.law					Salmonella sp.	None in 25g sample
					Enterobacteriaceae	≤ 1.0 x 10^3 UFC/g
					Fecal Streptococcus	≤ 1.0 x 10^3 MPN/g
					Nematodes	None in 50g sample
					Trematodes	None in 50g sample
					Cestodes	None in 50 g sample
The Netherlands[(z)]	55		4		Eelworms	None
BRL K256/02					Rhizomania virus	None
					Plasmodoph. Brass.	None
					Weeds	Germination ≤ 2 plant/l
UK Quality label[(x)]					*Salmonella spp*	*Absent in 25 g*
					E. coli	*1000 CFU g^{-1}*
					Weed propagules	*5 viable propagules l^{-1}*
					Plant tolerance	*20 % below control*

a) in vessel composting
b) open windrow composting
c) 2 approvals (1 in winter) for windrow composting
(x). taken from partner data
(z). water content depends on organic matter content
(q). Irish limits on maturity are divided into sets of options.

Table 15. Impurities Proportions In Compost Comparing Germany, Italy, The Netherlands And Austria

Country	*Impurities*	*Ø Mesh size*	*Limit Values % d.m. (m/m)*	
Austria Draft compost Ordinance	Total: agriculture	2 mm	≤ 0.5%	
	Total: land reclamation	> 2 mm	< 1%	
	Total: technical use	> 2 mm	< 2%	
	Plastics: agriculture	> 2mm	< 0.2%	
	Plastics: land reclamation	> 2mm	< 0.4%	
	Plastics: technical use	> 2mm	< 1%	
	Plastics: agric. excl. arable land	> 20mm	< 0.02%	
	Plastics: technical use	> 20mm	< 0.2%	
	Metals: agriculture	-	< 0.2%	
Belgium Flanders	Total	> 2mm	< 0.5%	
	Stones	> 5mm	< 2%	
Finland Fertil. Legislation	Total	-	< 0.5%	
France NF Composting of Municipal waste			**(A)**	**(B)**
	Plastics	> 5mm	< 0.5%	< 1.2%
	Heavy materials		< 6%	< 12%
	Total inert material		< 20%	< 35%
Germany Bio waste ordinance	Glass, plastics, metal	> 2mm	< 0.5%	
	Stones	> 5mm	< 5%	
Greece[(x)]	*Plastic*		*<0.3%*	
	Glass		*<0.3%*	
	Grading for 90% per weight	*< 10mm*		
Hungary[(x)]	*Glass*	*≥ 2mm*	*< 1%*	
	Plastic	*≥ 2mm*	*< 1%*	
	Metal	*≥ 2mm*	*< 0.5%*	
	Other inert substances (wood etc)	*≥ 2mm*	*< 1%*	
Ireland[(x)]	*Foreign matter*	*< 25mm*	*≤ 1.5%*	
Italy DPR 915/82	Total		≤ 3	
	Glass		≤ 3	
			≤ 1	
	Metals		≤ 0.5	
Fertil. Law	Plastics	< 3.33 mm	< 0.45%	
	Plastics	> 3,33 < 10mm	< 0.05%	
	Other inert materia	< 3.33mm	< 0.9%	
The Netherlands BOOM	Total	> 2mm	< 0.2%	
	Glass	> 2mm	< 0.2%	
	Glass	> 16mm	0	
	Stones	> 5mm	< 2%	
UK Quality label[(x)]	*Total glass, metal and plastic*	*> 2 mm*	*< 1% (of which < 0.5% is plastic)*	
	Stones and other consolidated mineral contaminants	*> 2 mm*	*< 5%*	

Table 16. Additional Limits

Country	Organic matter	Chloride (mg/kg d.m.)	EC (μS/cm)	Stability	C/N ratio	Oxygen uptake rate (O$_2$/kg volatile solids per hour)	pH
The Netherlands[x]	> 20% d.m.	< 5000	< 5.5	< 50°C			<6.5
Greece[x]							6-8
Ireland[x], [q]					< 25	< 50mg	

(x). taken from partner data
(q). Irish limits on maturity are divided into sets of options.

Table 17. Load Limits

Country		Phosphate (kg P_2O_5/ha)	Compost (tonnes/ha.y)
Netherlands[i], [ii]	Arable land	110	6 (12 in 2 years)
	Maize land	110	6 (12 in 2 years)
	Meadow	135	3 (6 in 2 years)
Other land (specific nature protection areas)		70	0

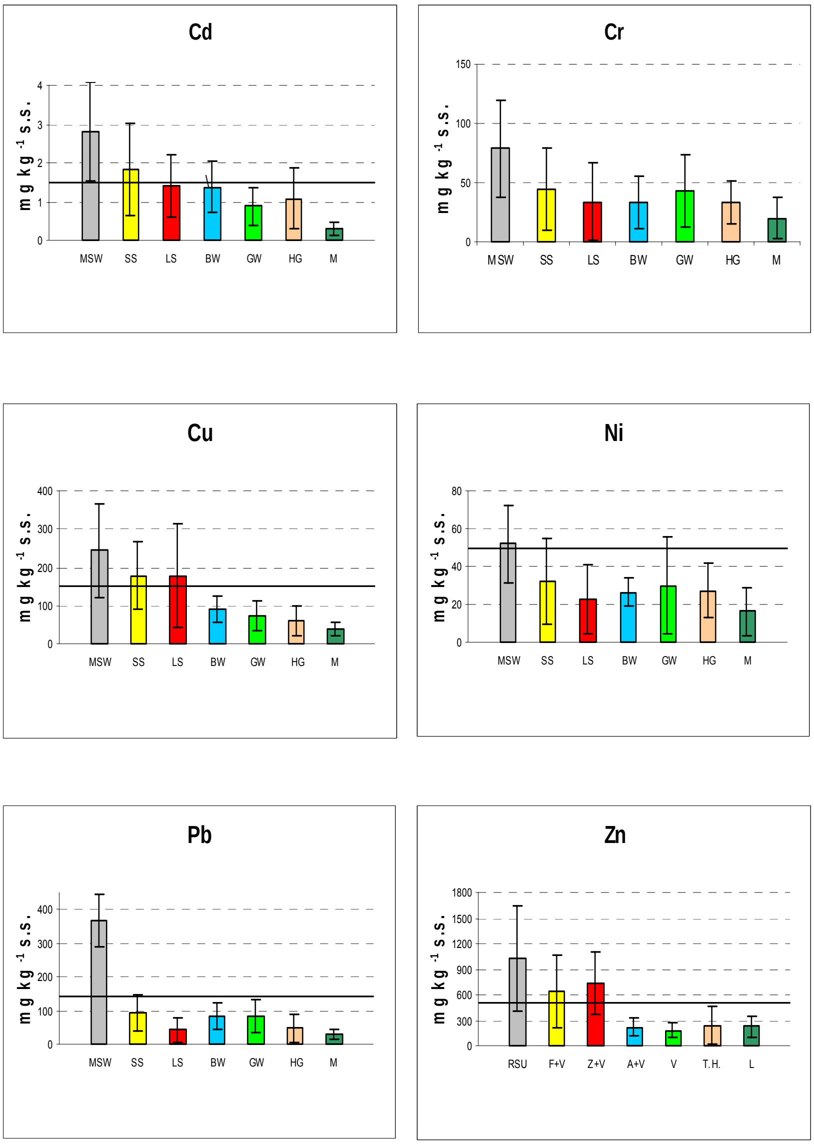
(i). phosphate loads apply to both compost and high quality compost
(ii). compost loads apply just to compost

Concentrating on compost standards in particular, and the factors associated with them, there are a number of important issues related to them forming a part of any proposed policy change. In facilitating the production of what can be considered as compost of a sufficient quality, it will be important to examine the feedstock and process, as well as the end-product. Furthermore, standards for compost can effectively draw the line between 'compost', which is a product, and 'waste', which is not. Waste is subject, under European law, to different legislation to that governing products.

Standards for Feedstock

For obvious reasons, the nature of the feedstock will have a large effect on the quality of the end-product. Feedstocks used in biological treatment processes include mixed municipal waste, source separated biowaste, sewage sludge, agro-industrial by-products, manure, slurries, yard waste, etc. Using source-separated biowaste, as opposed to mixed waste, results in a higher quality compost end-product with lower quantities of potentially hazardous materials (see Figure 12). In the absence of regulations concerning source separation, this raises the possibility of setting standards on the types of feedstock used in biological processes if the end-product is to be considered 'compost'.

Figure 12: Concentrations of Heavy Metals in Soil Improvers from Different Feedstocks Compared to Italian Fertiliser Law Limit Values



Municipal Solid Waste

Compost from Sewage Sludge (SS)

Compost from liquid slurries (LS)

Manure (M)

Compost from Biowaste (BW)

Compost from green waste (GV)

Soil Improvers for hobby gardening (HG)

Fertilisers Law (n° 748/84 mod. Decree 27.3.98)

Source: Centemero 2000

It can be seen from Figure 12 not only that the choice of feedstock used influences the concentration of heavy metals in the end product, but also that the effect of using compost derived from source separated biodegradable municipal wastes entails very little by way of changes in the loading of heavy metals. Indeed, the principal change which can be discerned from the Figure is the way in which using source-separated biowastes as the feedstock, as opposed to mixed municipal wastes, reduces the heavy metal content of the material. It should be said, however, that other impurities in mixed waste composts (glass, plastics, etc.) are as, if not more, important in reducing their desirability from the perspective of end users.

Setting standards on anything but the inclusion of broad categories of waste (including the examples given above), would be unrealistic as feedstock types are likely to vary considerably across the Member States. This gives rise to two main possibilities in terms of setting requirements on feedstock categories to be used in biological treatment processes:

1. specify categories of waste which can be used; or

2. specify categories of waste which should not be used.

Both alternatives will avoid the practices of 'mixing' and 'dilution' of wastes high in potentially hazardous materials, and will also ensure traceability of batches which produce sub-standard products (which is an important aspect of quality assurance). There are advantages and disadvantages to both possibilities. Specifying categories of waste which can be used will lead to a greater understanding of the feedstocks utilised and allow their more rigorous regulation. Specifying which feedstocks should not be used however will make possible the 'development' of feedstocks that would not be typically linked to composting, but subsequently could also provide a 'loop-hole' surrounding potentially undesirable wastes that do not fit into the prescribed categories. Enshrining such categories in law is also problematic in that if mistakes are made, the law has to be 're-opened', and this may be difficult to do without re-opening the space for complete re-negotiation of the law.

Standards for Process

The processes relating to biological treatment vary considerably across Europe. There are some characteristics which could be included in standards. An example is the compost curing time which will affect its maturity/stability. Irish regulations include stipulations under maturity that compost must be cured for at least 21 days; and it will not reheat upon standing to greater than 20^0C above ambient temperature, if it does not meet a number of other criteria.

Another parameter would be the temperature / time profile to which compost would be required to be subjected (this is usually specified as a certain minimum temperature to be reached for a minimum duration – see Table 14 above). This is specified to enable pasteurisation of the end product to ensure that certain pathogens, harmful either to humans or to livestock or plants, would be destroyed before compost was applied to land. It should also be remembered that the biological activity which 'makes' compost also has the effect of destroying pathogens (it is not just the pasteurisation effect which achieves this).

Other process related regulations that affect end-product quality might relate to the degree to which compost is aerated or turned. This will be important to ensure the composting process proceeded in the presence of sufficient air to avoid anaerobic activity.

Standards for End-products

Establishing standards on the end-products of biological treatment processes is generally accepted to be the most important area of regulation, especially since few ‘process-related’ standards can guarantee quality in the end product. Standards in the previous areas (feedstock and process) will, therefore, usually be complementary to standards on the end-product rather than substitutes for them. Only through tests on the end product can hygienisation and heavy metal limits be examined.

The main uses, in terms of market share, for compost (in descending order of importance) are: agriculture and special cultures, landscaping, hobby gardening, horticulture, earth works, landfill-restoration, and export. Each use has varying requirements, some similar and some different. The value of parameters such as maturity, conductivity, particle size, and nutrient content will be different across applications.

Examining national end-product standards across Europe, the most common ones limit the amount of potentially hazardous materials such as heavy metals and inert materials. Others include elements such as pathogens, which effectively act to support standards for the process (since the elimination of pathogens would be expected to follow from a well managed process), and the presence of weed seeds capable of germination.

As stated previously, compost has a number of end-uses. These include use in agriculture, landscaping, hobby gardening, horticulture, earth works, landfill-restoration, etc. Each use will have different requirements in terms of the quality of the compost required. For example, horticulture and agriculture will require higher standards than landscaping and landfill-restoration. From the end-user perspective, either statutory standards or voluntary quality assurance regimes can act as a mechanism to guarantee the quality of materials for specific end-uses. From the perspective of producers, standards act as a mechanism for segmenting markets, as well as a mechanism to ensure the higher value markets are not compromised by the marketing of low value products into markets for which they are unfit for purpose.

A number of countries, including Austria and the Netherlands, have set multiple standards for quality (class I & II; and ‘compost’ and ‘very good compost’ respectively). This raises the possibility of setting multiple standards according to end-use and/or quality. While this has the advantage of ensuring users obtain quality products, there are a number of complications including the fact that the local environments in the various member states will vary considerably, requiring different composts according to soil type, climate, etc. On the other hand, setting a single standards regime for compost across Europe would ease the process of regulation, and would facilitate marketing of compost (and composting / mechanical biological treatment processes) across national borders.

It may also be beneficial to set standards for maturity to distinguish between end-products that have gone through an extensive biological treatment process, stabilised materials, and those where the process produces a fresher product. Such standards may be especially important when considering the issue of stabilised biowastes arising from mechanical biological treatment. This is because the terms under which Member States may accept specific material in landfills, and / or the conditions under which waste treated through mechanical biological treatment may be considered 'no longer biodegradable' for the purposes of the Landfill Directive, have an important bearing upon the validity of such treatments and their 'competitiveness' in the market place for alternative waste treatments.

Standards to Protect Soil Quality

While setting limits on certain characteristics of compost is important, there are a number of other factors associated with compost use that need to be considered, including regulating its application to minimise negative environmental impacts. Examining existing standards at the EC level and across Member States, there are two main sets of limits, aside from simple composition constraints. These are:

- limit values for heavy metal concentrations in the soil (mg/kg d.m.)
- admissible loads of heavy metals (kg/ha.y)
- admissible loads for nutrients, such as nitrogen and phosphorous.

The last of these is often influenced by Member State implementation of the Nitrate Directive. It is worth pointing out that, particularly for composts from source-separated materials, the limits on application to soil are more usually reached in the context of limits on nitrate applications before limits on heavy metal loading are reached.

3.9 Summary

Key points that arise from this analysis of performance and standards are:

- In a number of countries, the source separation of biowastes is already far advanced. The role of composting in treating these materials is far more advanced than that of anaerobic digestion;
- Those countries / regions with the highest combined recycling / composting rates are those with systems in place for source-separation of compostable wastes. The contribution of this to landfill diversion is potentially considerable;
- These countries are also, in the main, those who have the most developed systems of standards for compost. This suggests an appreciation of the importance of these in developing end-use markets for compost; and
- However, there is not any 'agreed' system of standards and no obvious sign of 'clear convergence' towards some harmonised norm. This may reflect, in part, the different markets for end-products in the countries concerned, but it is also

likely to be a reflection of different regulatory approaches (and this reveals itself in the ‘coverage’ of the standards in place).

It would appear, therefore, that the source separation of biowastes has potential to impact considerably upon the generation of residual waste for recovery / disposal. As regards an approach to diversion of biodegradable municipal waste from landfill (as required under Article 5 of the Landfill Directive), it would appear to be a promising approach for those who have barely considered this route. Where countries take this approach seriously, the potential volume of compost generated makes it important to have in place a clear system of standards allowing the product to be properly marketed.

4.0 THE ECONOMIC ANALYSIS OF OPTIONS FOR MANAGING BIODEGRADABLE MUNICIPAL WASTE – EXTERNAL COSTS

4.1 Introduction

The approach adopted to assessing the external costs of the treatment options is principally 'top-down' in the sense that no dispersion modelling of atmospheric pollutants takes place. The environmental costs of different processes are addressed principally through quantitative measurements of emissions, these being multiplied by unit damage cost estimates to arrive at final figures for the externality.

There are a number of caveats that need to be borne in mind when considering the analysis:[15]

- As with most analyses of this nature, the external cost analysis is focused principally on air pollutants. Extension of the analysis has been made where possible, but the science underlying the impacts from emissions to other media is less clear (and potentially, less amenable to the marginalist analysis which characterises this type of analysis). Hence, there are a number of omissions from the analysis. These are made clear for each of the treatments;

- The impacts associated with plant construction have not been considered; and

- The use of unit damage costs to carry out the analysis effectively assumes that impacts are invariant with respect to the source of the emission. In reality, population density and the height from which emissions are discharged will have important effects on the actual external costs as would (in theory) the wealth of those located near to the plant (affecting willingness to pay to avoid the effects of pollution). Without more detailed modelling, making such adjustments is somewhat speculative. Separate analysis to assess transport effects has been carried out.

Partly for the reason just mentioned, ranges of estimates for the unit damage costs of specific pollutants have been used. In the case of greenhouse gases, the residence time of the gases is an important factor in determining their effect over time. The pure rate of time preference used (not, strictly speaking, the same as a discount rate) therefore affects the unit damage costs associated with these gaseous emissions. An account of the choice of unit damage costs is given in Appendix 1. For the most part, figures used by, or derived from studies by COWI (2000) and Brown et al (2000) have been used.

[15] For incineration and landfill, these are stated more fully in Broome et al (2000).

Whilst the following chapters include new work to understand the external costs and benefits of composting and anaerobic digestion, work on landfill and incineration is carried out principally through reference to existing work for Friends of the Earth and Waste Watch in the UK (Broome et al 2001) and by COWI (2000) for the European Commission, though with adaptation of emissions data used and unit damage costs applied.[16] No new work is carried out in, for example, consideration of gasification and pyrolysis technologies, which are already playing a role in strategies to meet Landfill Directive (LFD) targets. These are not considered further in this analysis.

The study does not take the approach, adopted in a number of studies, of linking a particular transport routing to a particular treatment option. This can lead to erroneous conclusions not least because:

- Both across and within different countries, the transport distances implied by all treatment options are likely to vary considerably;

- The transport distances are associated with the collection logistics which determine not just the vehicles used, but the nature of the materials collected and the frequency of their collection; and

- The transport-related externalities are not dependent on distance only. They will vary in accordance with the choice of transportation mode, the likelihood of accidents occurring and the location of any gaseous emissions (so, the routing of the journey undertaken). They also contribute to site-related disamenity affects.

For these reasons, transport related externalities are considered separately (see Section 4.6 below).

4.1.1 A Note on Climate Change Effects

The way in which climate change is treated in studies of this nature is deserving of some discussion. This is usually treated in such a way as total emissions over time are compared across the waste management options. Furthermore, the effect of carbon dioxide emissions from biogenic sources is usually treated as a 'neutral process' since the 'baseline assumption' is a situation in which the material would have degraded at ambient temperatures.

This is a flawed approach since it ignores the time dimension in understanding the flux of greenhouse gases into the atmosphere. Furthermore, the impacts of different greenhouse gases change in their relative economic significance as the discount rate applied is changed. This is because the gases have different residence times in the atmosphere.

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[16] Other relevant studies are PIRA et al (1998), Rabl et al (1998), AEA Technology (1997) Brisson and Pearce (1995), Brisson and Powell (1995), Powell et al (1995) and Powell et al (1996).

For this reason, a more sophisticated treatment must look at the time profile of these (and all other emissions). This applies to the biogenic greenhouse gas emissions as well as the non-biogenic ones (and as far as possible, it ought to apply to all impacts). In this analysis, therefore, there is no ‘subtraction’ of the ‘carbon dioxide emissions which would have occurred anyway’ if the material degraded aerobically over time (other studies use the distinction between ‘short-‘ and ‘long-term’ carbon). Of principal concern here is the comparative analysis of the options, not the measurement of an inventory of emissions against pre-established criteria. Furthermore, these inventories, assessed on a year-by-year basis, ought to allocate emissions to the year in which they are produced. It is quite clear that from this perspective, the emissions from processes, such as incineration, which occur immediately, should be treated differently from those of compost and landfill, which occur over more protracted periods of time.

Once one appreciates the time dimension (and this may be significant in the context of decisions regarding climate change emissions, and in terms of their valuation at different rates of discount), one begins to appreciate the limitations of setting to one side emissions from biogenic sources. Table 18 below shows a comparison of assumptions made in this study with those that have typically been made in past studies. These assumptions imply a more ‘correct’ comparison across options, certainly from the perspective of appreciation of, and valuation of, environmental impacts at different rates of discount.

Table 18. Comparison of Assumptions Made Affecting Treatment of Greenhouse Gas Emissions Relative to ‘Standard’ Assumptions

Key Assumption	Landfill	Incineration	Composting
Typical Life-cycle approach: Emissions of carbon dioxide from biogenic sources can be ignored	Emissions of Methane are calculated. Emissions of carbon dioxide are ignored. Where methane is captured and flared / used for energy recovery, the assumption implies there is no net greenhouse gas effect. All emissions are effectively treated as though they occur instantaneously.	Only carbon dioxide from non-biogenic sources (and nitrous oxide, where this is considered) are assessed. Where there are assumptions made concerning displaced energy sources, these emissions are offset against emissions which would have occurred had the energy been generated using another source.	All emissions are assumed to be of biogenic origin, therefore as long as the process is entirely aerobic, there are no ‘net greenhouse gas emissions.’ Where emissions of methane are assumed, these are accounted for. There may be accounting for avoided emissions from fertiliser manufacture.
This study: Emissions of all greenhouse gases are accounted for and allocated to the year in which they occur	Emissions of all greenhouse gases are accounted for and are allocated to the year in which they occur. The economic impacts are then evaluated at different discount rates and the net present value of the impact of these emissions is calculated.	All greenhouse gas emissions from incineration, including N_2O, are assessed, irrespective of the nature of the material incinerated. Different assumptions concerning energy displacement are considered in respect of offsetting greenhouse gas credits.	Emissions from the process are considered first. However, when the product is ‘used’, carbon is returned to the soil. There, it is progressively mineralised over time. All emissions are allocated (both those from the process, and those post-application to the soil) over time, as well as the effect in terms of fertiliser displacement etc. as this occurs in time.

4.1.2 A Note on Energy Displacement

The recovery of energy from waste is preferable to one where energy is generated without any recovery of the resulting energy. How should that benefit be accounted for in environmental terms, if at all (should the benefits be viewed solely in terms of an additional private benefit from the recovery process)?

The approach usually adopted in studies of this nature is to assume that energy from waste 'displaces' emissions which would be derived from another energy source if it were used to supply the equivalent energy. The process is then credited with these 'avoided emissions'.

But which energy source is being 'displaced' by these processes? The question is more than merely academic since all studies indicate that the effect of this assumption is absolutely critical in the analysis of external costs of those waste treatments which recover energy.

Typically, a marginalist approach is adopted. Hence, for example, in recent work undertaken for the Commission,

electricity and heat recovery will displace the least profitable form of electricity generation in the electricity system, which means that the recovery of energy will displace pollution from those sources. This gives rise to external benefits for society.

[…]the external benefit of displaced emissions depends on the energy source that the energy replaces. […]

When valuing the benefit of displaced emissions, emphasis should be placed on determining/assuming the marginal source of both the heat and the power replaced. In CSERGE et al (1993), EC (1996d) and Brisson (1997) the marginal source of power is assumed being coal-fired power stations. This is still a common source to production of primary energy in several European countries. Coal-fired power stations are very polluting and therefore this assumption results in significant external benefits from displaced emissions. However, EC (1996d) and Brisson (1997) also present an alternative scenario assuming an average European fuel mix. To choose the right energy source replaced in actual calculations, the current and the likely future marginal source of power should be reviewed for the specific country.

Sometimes it is argued that it is not always correct to assume that the marginal source of power is coal-fired power stations. It is argued that the alternative to using energy from incineration of waste could very well be energy from new, efficient power plants. However, the marginal alternative should always be assessed in the short term from the existing power plants, in which case coal-fired power stations are the right marginal source in most EC countries (COWI 2000).

It is debatable whether this analysis is strictly correct. First of all, even if the marginalist approach were the correct one, the fact that different energy sources are used at on- and off-peak times in many countries would require one to allocate, somewhat absurdly, different external benefits depending upon the time at which a particular tonne of waste had energy derived from it. In this case, the average source might be a more appropriate choice upon which to base the assessment of benefits of energy recovery.

However, the marginalist analysis warrants closer scrutiny still. Most thermal treatment plants are constructed on the basis that, as far as possible, they will run at full capacity. It is only to a limited degree, therefore, that any switches *at the margin* occur in which waste which was not destined for the plant becomes so, or vice versa. There are few 'marginal tonnes' of waste in this context.

In this case, though, does the plant really 'displace' any existing energy source at all, or is it simply helping to meet increasing demand for energy? This is an empirical question, but projections suggest that demand for energy is increasing. Within the EU, electricity demand is projected to grow at 1.8% per annum between 1997 and 2010 (from 2,422 TWh to 3,058 TWh).[17] Consequently, the increase in electricity supplied from renewable energy (250TWh from 1997 to 2010) will not be sufficient to reduce the total supplied from non-renewable sources, which will continue to grow (by 636 TWh from 1997 to 2010). There is, therefore, no 'displacement' of non-renewable sources as such. Renewables and non-renewables alike will simply contribute to meeting projected increases in demand.

Where energy from waste is used for district heating, it also becomes somewhat contentious to assume that the displacement is always of the marginal source for electricity generation. It could be argued that even the marginalist assumption should look more closely at the source of heat energy (as opposed to electricity). More controversially, some might claim that the CHP plant, to the extent that it displaces any source of energy, is displacing 'other CHP plant'. Hence, if a district heating system is in place, the displaced source might be taken to be an alternative CHP supply to the district heating system (because the infrastructure itself is likely to be used by a similar plant if the energy from waste plant did not exist).

These points are made all the more pertinent by virtue of the emerging policy position in Europe. Targets are being set for all Member States for the proportion of electricity to supply to be met by renewable sources by 2010. Energy from waste is currently included within the scope of that target. Arguably, in this policy environment, if these targets are just met, the net effect of an increase in energy from waste capacity is a reduction in the requirement for 'other renewables' which contribute to the target. Hence, it may even be argued that energy from waste is, in this policy context, displacing, at the margin, other renewable sources of energy.[18]

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[17] Figures taken from Energy Outlook (1997 figure) and the Proposal for the Directive (2010 figure).

[18] Note that the substance of this argument would be altered were it to be accepted that energy from waste is not a renewable source of energy. As mentioned above, the view that the very substance one is trying to minimise – waste – should be considered a renewable source of energy seems illogical. It also risks sending perverse messages to the public of the nature that it is acceptable to generate waste because 'waste is a renewable source of energy'. Generating energy from waste makes sense if one must combust it, but that does not qualify the energy as being from a renewable source.

The significance of this discussion is that, depending upon the viewpoint adopted (and it seems unlikely that there will be agreement as to which assumption should be carried forward because of the different interests involved), the 'avoided emission' credited on the basis of energy produced from waste can change radically.

In this study, two assumptions are carried forward:

- The first is that energy from waste acts to displace sources of energy which would otherwise have to be built (because demand for energy is, disconcertingly, one might say, increasing). Therefore, the source of energy being displaced is assumed to be one representative of new build energy. In this study, these are taken to be combined cycle gas turbines (CCGT). This is the same assumption used in recent work on PVC Waste Management (Brown et al 2000); and
- The second is that no source of energy is being displaced, which is broadly equivalent to an assumption that one is displacing a clean source with little or no disamenity impact (such as, for example, photovoltaics). It could be argued that new renewable energy targets are likely to require that significant proportions of new-build will be renewable sources. This seems to be the best alternative assumption to the one used above, and represents the desired effect of changing policies in this area.

Note that under the first assumption, where energy recovery is from combined heat and power plant, it would not be strictly correct to allocate the same 'avoided burdens' from the heat recovery as from the electricity recovery. For the gas fuels cycle in the Danish case under the ExternE programme, the external costs from heat generation via CHP were allocated on the basis of exergy. This led to externalities per kilowatt hour from heat energy generation being approximately a quarter those from the generation of electricity. In the German case study, the same approach led to the heat energy externalities being two-thirds of those from electricity generation. In this study, an assumption is made that for CHP plant, where burdens are assumed to be avoided, the avoided burdens per unit of thermal energy generation are one half those from electricity generation.

4.1.3 Avoided External Cost Data

In terms of the treatment of avoided external costs of energy, in order to give the analysis greater relevance at the country specific level, country-specific estimates of external costs of energy supply, taken from the ExternE national implementation studies, have been used for the 'avoided energy' assumption (European Commission 1999a).

The approach taken has been to use the figures for the externality per kWh and per GJ associated with electricity generation from combined cycle gas fired power stations in the countries concerned. For Sweden, Ireland and Finland, since no results for the gas cycle are reported in those countries, the figures used are 35% of the values reported for the coal-fired power station (this relationship holds approximately true for the other Member States reporting on both cycles) (see Table 19). The ExternE results have been updated through applying European GDP inflators to the 1995 values. Since the whole ExternE valuation process was conducted in Euros (and the country specific variations relate to factors such as population density etc.) no account has been taken of exchange rate movements. However, an inflation of the valuation of life has been assumed to reflect increases in real GDP.

Table 19. External Costs Associated With Electricity Production from Gas In Different Countries

		AUS	BEL	DEN	FIN	FRA	GER	GRE	IRE	ITA	LUX	NL
€ per MWh (1995)	*Low*	11	11	15	7	24	12	7	20.65	15	5	5
	High	26	22	30	15.4	35	23	13	29.4	27	19	19
€ per GJ (1999)	*Low*	3.46	3.46	4.72	2.20	7.55	3.78	2.20	6.50	4.72	1.57	1.57
	High	8.18	6.92	9.44	4.85	11.01	7.24	4.09	9.25	8.50	5.98	5.98
		POR	**SPA**	**SWE**	**UK**	**CYP**	**CZ**	**EST**	**HUNG**	**POL**	**SLO**	**EU-15 AVE**
€ per MWh (1995)	*Low*	8	11	6.3	11							
	High	21	22	14.7	22							
€ per GJ (1999)	*Low*	2.52	3.46	1.98	3.46	3.54	3.54	3.54	3.54	3.54	3.54	3.54
	High	6.61	6.92	4.63	6.92	7.10	7.10	7.10	7.10	7.10	7.10	7.10

Source: European Commission (1999a)

Note that these figures have their own shortcomings. First of all, they are based on old data (so they may overstate the external costs if technologies for generation have improved). Secondly, they are based upon analysis carried out using the 3% discount rate as the central assumption (though the high and low ranges typically span discount rates of 1% and 3%). There is no available basis for varying these externalities as the discount rate changes (though the 3% case is the central assumption in this analysis – see next section). Lastly, regarding the valuation of health effects, although the ExternE approach looked at use of both the 'value of statistical life' and the 'years of life lost' approaches to placing values on mortality, most of the ranges reported reflect the latter approach. Given that there is still much debate about the appropriate approach, some may say this underestimates the effects being valued (since using the value of statistical life generates higher external cost estimates) (European Commission 1999b).

For the Accession States, the EU average has been used. There is a question as to whether this is valid. Two key criticisms of this approach could be advanced:

a) the plant used to generate electricity are likely to be 'dirtier'; and

b) the willingness to pay for avoidance of pollution is not as high, reflecting the lower purchasing power of per capita GDP.

On the other hand, Accession States will, in due course, be required to comply with Directives affecting, for example, both air quality in general, and emissions from large combustion plant.

On those grounds, one could, over the medium- to long-term, work with EU averages on the assumption that similar emissions standards will be reached by power plants in Accession States. With regard to the willingness to pay issue, for the majority of the Accession States, the purchasing power of per capita GDP is clearly lower than in the majority of EU Member States. However, there is considerable variation within the EU also. The ExternE research from which these figures are derived appear not to account for the relative PPP adjusted incomes in the different Member States to calculate the external costs of electricity supply (for example, by adopting what would no doubt have been a controversial stance in using different values for statistical life and years of life lost). To this end, the EU average is used, though there are clearly reasons why some might argue one should do otherwise.

These figures are used in all of the treatment options which involve assessment of the avoided external costs of electricity supply (including where processes use electrical energy).

4.1.4 Discounting and 'Country-specific' Externalities

The discounting of future benefits is always significant in any economic analysis. The study attempts to introduce rather more of a time dimension into this analysis than in earlier studies. Those earlier studies have implicitly, by valuing all emissions in the life-cycle at the same level even when they occur over protracted periods of time, employed a zero rate of discount. The approach in this study is different.

It is usual in analyses for the European Commission to work with discount rates of 2%, 4% and 6%. The choice of discount rates which can be modelled has effectively been dictated by the discount rates for which damages associated with greenhouse gases have been estimated, that is, 1%, 3% and 5%. It will be increasingly important in future analyses to recognise the significance of discounting and the effects this has on the valuation of global warming externalities. This is more important in terms of understanding the 'relative' effects rather than the absolute ones, since any attempt to value marginal damages associated with incremental emissions of greenhouse gases must itself be treated with extreme caution (there is, after all, not so much certainty about the magnitude of the effects which should be valued).

The ExternE work (from which country-specific energy externalities are taken) was also used as a source of country-specific unit damage costs for SO_2, NO_x and particulates (European Commission 1999a). The ExternE work used the 3% discount rate as the central assumption, using 1% and 10% for sensitivity analysis. This suggests that the 3% discount rate is likely to produce the most reliable results, and it is, perhaps, in any case, the most realistic result to use. Arguably, if one varies the discount rate (and with it, the global warming unit damage costs), one ought also to vary the unit damage costs associated with any of the other pollutants which have an effect which extends across time. There is no basis available for making such adjustments, since the impacts of other pollutants (such as they are known) prevent one from making such adjustments (where they might be necessary). Since the ExternE programme did not look at Accession States (it is now doing so), no specific estimates for these countries exist. A wider range of valuation figures is used for Accession States based upon the ranges across the EU Member States.

The other reason for favouring the 3% discount rate is that the avoided energy externalities were calculated using this discount rate as the central assumption. Consistency demands that the damages / avoided damages associated with energy use / production should be varied as the discount rate changes. In this analysis, this has not been possible so the 3% figure is the only one which is 'consistent' with estimates of externalities from energy generation. That is why in the generation of the results, the 3% discount rate assumption is given far greater significance.

4.2 Results for Landfill

The details of the analysis are given in Appendix 2. Table 20 shows which external costs are covered and which are not in this analysis. It is quite clear from this that many externalities which one would assume to be negative have not been quantified in the analysis. As such, this almost certainly constitutes an under-estimate of the external costs associated with landfill.

Table 20. External Costs of Landfill Covered In the Analysis

Category	Covered	Not Covered
Air Emissions	Methane, Carbon dioxide, NO_x, CO_2, N_2O, CO, VOCs, dioxins, cadmium, chromium, lead, particulates[19]	All other air emissions, including H_2S, HCl, HF, mercury, zinc etc.
Emissions to Water	Leachate (in a limited fashion)	
Other	Disamenity	Accidents (fires, explosions, groundwater contamination, etc.) Operator health issues (related to waste handling / movements, respiratory effects) Plant (construction) related externalities On-site fuel use Non-market benefits from recreational uses post-closure (to the extent that one accepts such benefits can be attributed to the process) Unproven effects (such as possible birth defects)
Avoided burdens from energy generation	Air emissions from CHP plant (covering various phases of extraction to generation, but restricted to air pollutants)	Other external costs of extraction etc. as well as construction of plant etc.

For the purpose of this analysis, the important results are those to do with external costs avoided (i.e. the external benefits generated) when a tonne of material collected at source is removed from landfill. This analysis is shown for two cases, both using the central (3%) discount rate.

[19] These were valued as PM_{10}. Some studies use higher unit damage costs than for PM_{10} on the basis that particulates from incinerators are expected to be the sub-2.5 micron type, and these are believed to be more damaging to human health (see, e.g., PIRA et al 1998).

In the first case, the results are shown using estimates for the gas collection efficiency of landfills based on Smith et al (2001). Smith et al (2001) suggest as of 2000, 90% of landfills in the UK had gas control systems in place, whilst figures for Austria and Spain were 33% and 23% respectively. Table 21 shows estimates for 'country-specific' gas collection efficiencies based largely on those in Smith et al (2001). Table 22 then shows the external benefits of reducing, by one tonne, the amount of biowaste landfilled using these country-specific estimates. In all cases, it was assumed that 50% of the gas collected is used to generate energy, the efficiency of the engine being 30%. Of the uncollected gas, 10% is assumed to be oxidised either at the cap or in leachate. Appendix 2 Illustrates how sensitive the analysis is to these assumptions. Note that individual landfills may perform better or worse than these figures suggest.

Table 22 reports the total external benefit from reducing landfilling of biowaste from landfill using different assumptions about disamenity and energy displacement. The reason for showing the different results (i.e. with and without disamenity and avoided burdens from energy generation) is to illustrate the significance of the assumptions regarding energy recovery and disamenity. In the case of landfill, there is relatively little effect since:

- the energy recovered is relatively small and it is recovered over a period of time (so that the discounted flow of benefits is relatively small); and
- the population density around the landfill is typically low and so the unit disamenity-related externality is small also (since these are effectively calculated through a hedonic pricing equation relating disamenity to changes in house prices).

As Appendix 2 shows, the more important variables, in absolute terms, are the discount rate used (because of the time profile for gas generation) and the parameters defining landfill performance (such as efficiency of gas capture and energy generation).

Given that, later in the analysis, one seeks to understand the likely evolution of waste management in Member States, estimates have also been made for the external benefits of taking biodegradable waste away from landfill using, for all countries, a collection efficiency of 60%. This has the effect of reducing the external benefits of removing waste from landfill. This is deemed broadly representative of the situation which might prevail in 2010, by which time, implementation of the technical aspects f the Landfill Directive will be complete. This requires landfills to have gas collection equipment in place, for recovery where possible. The results are shown in Table 23.

Table 24 illustrates the differences in the external benefits under the 'current' (Table 21) and 'with Landfill Directive' (Table 23) scenarios. The fact that the numbers are negative indicates that, as would be expected, the benefits from removing waste from landfill will fall as the operation of landfills improves in the European Union. The numbers are greatest for those countries whose landfills are assumed to be furthest from good practice under the assumptions made in Table 21.

It should be emphasised that in the landfill case, as with all other treatments, the benefits assessment for landfill covers a sub-set of the total externalities. It is almost certainly the case that the externalities for landfill are underestimates.

Table 21. Estimates of Country-specific Landfill Gas Capture Rates

	AU	BE	DK	FI	FR	GE	GR	IR	IT	LU	NL	PO	SP	SW	UK	CYP	CZ	EST	HUN	POL	SLO
% of waste in landfill sites with gas control	33%	60%	90%	90%	60%	90%	10%	23%	60%	60%	90%	23%	23%	90%	90%	10%	10%	10%	10%	10%	10%
Methane collection efficiency	20%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	50%	63%	50%	50%	50%	50%	50%	50%	50%	50%
Country gas collection efficiency	6.6%	30.0%	45.0%	45.0%	30.0%	45.0%	5.0%	11.5%	30.0%	30.0%	45.0%	11.5%	14.5%	45.0%	45.0%	5.0%	5.0%	5.0%	5.0%	5.0%	5.0%

Source: Based on Smith et al (2001). The only estimate which has been changed is that for the UK, where the figure of 63% for capture of all methane from landfills seems too high. This has been reduced to the same level as Germany, Netherlands etc.

Table 22. Effects of Reducing Quantity of Biowaste Landfilled by One Tonne (€ / tonne, using country-specific methane captures)

3% Discount Rate	AU		BE		DE		FI		FR		GE		GR		IR		IT		LU		NL	
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Net External Cost (air pollutants / climate change) = A	22.00	24.82	18.3	20.65	15.80	17.94	15.78	17.9	18.27	20.74	15.8	18.06	22.08	24.91	21.09	23.71	18.21	20.66	18.2	20.57	15.84	17.97
Avoided Burdens (Energy Production) = B	-0.08	-0.19	-0.36	-0.71	-0.73	-1.46	-0.34	-0.75	-0.78	-1.14	-0.58	-1.12	-0.04	-0.07	-0.26	-0.37	-0.49	-0.88	-0.16	-0.62	-0.24	-0.93
Disamenity = C	2.04	4.08	7.13	14.25	2.98	5.96	0.35	0.71	2.21	4.42	5.33	10.67	1.38	2.75	0.88	1.75	3.65	7.29	3.35	6.71	8.48	16.96
Net Externality (excl. disamenity, incl. displaced burdens from energy generation) = A+B	21.92	24.63	18	19.94	15.10	16.47	15.43	17.15	17.49	19.6	15.3	16.94	22.05	24.84	20.83	23.34	17.72	19.79	18.1	19.95	15.6	17.04
Net Externality (excl. disamenity, excl. displaced burdens from energy generation) = A	22.00	24.82	18.3	20.65	15.80	17.94	15.78	17.9	18.27	20.74	15.8	18.06	22.08	24.91	21.09	23.71	18.21	20.66	18.2	20.57	15.84	17.97
Net Externality (incl. disamenity, incl. displaced burdens from energy generation) = A+B+C	23.96	28.71	25.1	34.19	18.10	22.43	15.78	17.86	19.7	24.02	20.6	27.61	23.43	27.59	21.71	25.09	21.37	27.08	21.4	26.66	24.08	34
Net Externality (incl. disamenity, excl. displaced burdens from energy generation) = A+C-B	24.04	28.9	25.4	34.9	18.80	23.90	16.13	18.61	20.48	25.16	21.2	28.73	23.46	27.66	21.97	25.46	21.86	27.95	21.6	27.28	24.32	34.93
	PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO			
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High		
Net External Cost (air pollutants / climate change) = A	21.14	23.75	20.6	23.33	15.80	17.91	15.84	18.01	22.17	24.94	22.2	24.94	22.17	24.94	22.17	24.94	22.17	24.94	22.2	24.94		
Avoided Burdens (Energy Production) = B	-0.10	-0.26	-0.17	-0.35	-0.31	-0.72	-0.54	-1.07	-0.06	-0.12	-0.06	-0.12	-0.06	-0.12	-0.06	-0.12	-0.06	-0.12	-0.06	-0.12		
Disamenity = C	1.85	3.71	1.27	2.54	0.56	1.13	5.21	10.42	1.254	2.508	2.03	4.053	0.52	1.039	1.661	3.322	1.935	3.87	1.5	3.008		
Net Externality (excl. disamenity, incl. displaced burdens from energy generation) = A+B	21.04	23.49	20.5	22.99	15.50	17.19	15.30	16.94	22.11	24.82	22.1	24.82	22.11	24.82	22.11	24.82	22.11	24.82	22.1	24.82		
Net Externality (excl. disamenity, excl. displaced burdens from energy generation) = A	21.14	23.75	20.6	23.33	15.80	17.91	15.84	18.01	22.17	24.94	22.2	24.94	22.17	24.94	22.17	24.94	22.17	24.94	22.2	24.94		
Net Externality (incl. disamenity, incl. displaced burdens from energy generation) = A+B+C	22.89	27.2	21.7	25.53	16.00	18.32	20.51	27.36	23.36	27.32	24.1	28.87	22.63	25.86	23.77	28.14	24.04	28.69	23.6	27.82		
Net Externality (incl. disamenity, excl. displaced burdens from energy generation) = A+C-B	22.99	27.46	21.9	25.87	16.40	19.04	21.05	28.43	23.42	27.45	24.2	28.99	22.69	25.98	23.83	28.26	24.1	28.81	23.7	27.95		

Note: Low and High figures refer to the use of 'sets' of low and high unit damage costs for the various pollutants under consideration. Positive numbers indicate external benefits of reducing landfilled material by one tonne of biowaste.

Table 23. Effects of Reducing Quantity of Biowaste Landfilled by One Tonne (€ / tonne, good practice in all countries)

3% Discount Rate	AU		BE		DE		FI		FR		GE		GR		IR		IT		LU		NL	
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Net External Cost (air pollutants / climate change) = A	13.49	15.45	13.52	15.42	13.44	15.35	13.42	15.32	13.49	15.47	13.47	15.44	13.42	15.38	13.43	15.34	13.45	15.43	13.46	15.37	13.46	15.37
Avoided Burdens (Energy Production) = B	-0.71	-1.69	-0.71	-1.43	-0.97	-1.95	-0.45	-1.00	-1.56	-2.27	-0.78	-1.49	-0.45	-0.84	-1.34	-1.91	-0.97	-1.75	-0.32	-1.23	-0.32	-1.23
Disamenity = C	2.04	4.08	7.13	14.25	2.98	5.96	0.35	0.71	2.21	4.42	5.33	10.67	1.38	2.75	0.88	1.75	3.65	7.29	3.35	6.71	8.48	16.96
Net Externality (excl. disamenity, incl. displaced burdens from energy generation) = A+B	12.78	13.77	12.80	13.99	12.46	13.40	12.96	14.32	11.93	13.20	12.69	13.95	12.97	14.53	12.09	13.43	12.48	13.67	13.14	14.14	13.14	14.14
Net Externality (excl. disamenity, excl. displaced burdens from energy generation) = A	13.49	15.45	13.52	15.42	13.44	15.35	13.42	15.32	13.49	15.47	13.47	15.44	13.42	15.38	13.43	15.34	13.45	15.43	13.46	15.37	13.46	15.37
Net Externality (incl. disamenity, incl. displaced burdens from energy generation) = A+B+C	14.82	17.85	19.93	28.24	15.44	19.36	13.31	15.03	14.14	17.62	18.02	24.62	14.35	17.28	12.97	15.18	16.13	20.96	16.49	20.85	21.62	31.10
Net Externality (incl. disamenity, excl. displaced burdens from energy generation) = A+C-B	15.53	19.53	20.65	29.67	16.42	21.31	13.77	16.03	15.70	19.89	18.80	26.11	14.80	18.13	14.31	17.09	17.10	22.72	16.81	22.08	21.94	32.33
	PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO			
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High		
Net External Cost (air pollutants / climate change) = A	13.46	15.36	13.45	15.41	13.43	15.33	13.46	13.46	15.39	13.46	15.39	13.46	15.39	13.46	15.39	13.46	15.39	13.46	15.39	15.39		
Avoided Burdens (Energy Production) = B	-0.52	-1.36	-0.71	-1.43	-0.41	-0.95	-0.71	-0.73	-1.47	-0.73	-1.47	-0.73	-1.47	-0.73	-1.47	-0.73	-1.47	-0.73	-1.47	-1.47		
Disamenity = C	1.85	3.71	1.27	2.54	0.56	1.13	5.21	1.25	2.51	2.03	4.05	0.52	1.04	1.66	3.32	1.94	3.87	1.50	3.01	6.22		
Net Externality (excl. disamenity, incl. displaced burdens from energy generation) = A+B	12.94	14.00	12.73	13.98	13.02	14.37	12.75	12.73	13.92	12.73	13.92	12.73	13.92	12.73	13.92	12.73	13.92	12.73	13.92	13.92		
Net Externality (excl. disamenity, excl. displaced burdens from energy generation) = A	13.46	15.36	13.45	15.41	13.43	15.33	13.46	13.46	15.39	13.46	15.39	13.46	15.39	13.46	15.39	13.46	15.39	13.46	15.39	15.39		
Net Externality (incl. disamenity, incl. displaced burdens from energy generation) = A+B+C	14.79	17.71	14.00	16.52	13.58	15.50	17.96	13.98	16.43	14.75	17.98	13.24	14.96	14.39	17.25	14.66	17.79	14.23	16.93	20.15		
Net Externality (incl. disamenity, excl. displaced burdens from energy generation) = A+C-B	15.31	19.07	14.72	17.95	13.99	16.46	18.67	14.71	17.90	15.48	19.44	13.98	16.43	15.12	18.71	15.39	19.26	14.96	18.40	21.61		

Note: Low and High figures refer to the use of 'sets' of low and high unit damage costs for the various pollutants under consideration. Positive numbers indicate external benefits of reducing landfilled material by one tonne of biowaste.

Table 24. Change in External Benefit From Reducing Landfilling of Biowaste by One Tonne Implied by Move to Good Practice Landfilling (€/tonne)

3% Discount Rate	AU		BE		DE		FI		FR		GE		GR		IR		IT		LU		NL	
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Net Externality (excl. disamenity, incl. displaced burdens from energy generation) = A+B	-9.14	-10.86	-5.15	-5.95	-2.61	-3.08	-2.47	-2.83	-5.56	-6.41	-2.58	-3.00	-9.08	-10.31	-8.74	-9.91	-5.24	-6.11	-4.92	-5.81	-2.46	-2.91
Net Externality (excl. disamenity, excl. displaced burdens from energy generation) = A	-8.51	-9.36	-4.80	-5.23	-2.37	-2.59	-2.36	-2.58	-4.78	-5.27	-2.38	-2.62	-8.66	-9.53	-7.65	-8.37	-4.75	-5.24	-4.76	-5.20	-2.38	-2.60
Net Externality (incl. disamenity, incl. displaced burdens from energy generation) = A+B+C	-9.14	-10.86	-5.15	-5.95	-2.61	-3.08	-2.47	-2.83	-5.56	-6.41	-2.58	-3.00	-9.08	-10.31	-8.74	-9.91	-5.24	-6.11	-4.92	-5.81	-2.46	-2.91
Net Externality (incl. disamenity, excl. displaced burdens from energy generation) = A+C-B	-8.51	-9.36	-4.80	-5.23	-2.37	-2.59	-2.36	-2.58	-4.78	-5.27	-2.38	-2.62	-8.66	-9.53	-7.65	-8.37	-4.75	-5.24	-4.76	-5.20	-2.38	-2.60
	PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO			
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High		
Net Externality (excl. disamenity, incl. displaced burdens from energy generation) = A+B	-8.10	-9.49	-7.74	-9.01	-2.47	-2.82	-2.56	-2.97	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89		
Net Externality (excl. disamenity, excl. displaced burdens from energy generation) = A	-7.68	-8.39	-7.20	-7.92	-2.36	-2.58	-2.38	-2.61	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55		
Net Externality (incl. disamenity, incl. displaced burdens from energy generation) = A+B+C	-8.10	-9.49	-7.74	-9.01	-2.47	-2.82	-2.56	-2.97	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89	-9.38	-10.89		
Net Externality (incl. disamenity, excl. displaced burdens from energy generation) = A+C-B	-7.68	-8.39	-7.20	-7.92	-2.36	-2.58	-2.38	-2.61	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55	-8.71	-9.55		

It is important to appreciate (with the total cost benefit analysis in mind) that some countries operate policies which enable electricity generated from landfill gas to be sold at a price above the prevailing market rate for electricity. This means that one must be very careful in attributing environmental benefits to the energy generated from landfills in the context of a full cost benefit analysis. To the extent that some countries do offer subsidies, it can be argued that these benefits are already internalised in the subsidy regime so that they are reflected in private costs (see next Chapter).

Similarly, some countries have taxes on landfill in place. To the extent that these reflect the external costs of landfilling, the taxes ought not to be considered in isolaton from the external cost analysis since this would imply double counting of the environmental costs of landfilling. Similar considerations apply with respect to incineration (see Section 4.3).

Lastly, for certain countries, the landfill option may become more or less irrelevant in years to come. Several countries have, or intend to have, bans in place for landfilling of municipal waste, or specific fractions of municipal waste, or municipal waste which has not been pre-treated according to specific requirements.

4.3 Results for Incineration

Full details are given in Appendix 3. Table 25 shows which external costs are covered and which are not in this analysis.

Table 25. External Costs of Incineration Covered In the Analysis

Category	Covered	Not Covered
Air Emissions	SO_x, NO_x, CO_2, N_20, CO, VOCs, dioxins, cadmium, chromium, lead, arsenic, particulates[20]	All other air emissions, including HCl, HF, mercury etc. Exceedences of limit values Consistent 'beyond Directive' performance
Emissions to Water	Assumed to be to some degree internalised through costs of waste water treatment (so not explicitly covered)	
Emissions to Land	Avoided external costs of aggregates extraction associated with use of bottom ash in construction	Treatment / landfilling of fly-ash Treatment of bottom ash prior to utilisation
Other	Disamenity	Accidents (fires, etc.) Operator health issues (related to e.g. ash handling) Primary resource extraction (related to water use and minerals used in flue gas treatment) Plant (construction) related externalities (including those related to district heating networks) On-site fuel use Transport of ash residues to final destinations

[20] These were valued as PM_{10}. Some studies use higher unit damage costs than for PM_{10} on the basis that particulates from incinerators are expected to be the sub-2.5 micron type, and these are believed to be more damaging to human health.

Avoided burdens	Air emissions from CHP plant (covering various phases of extraction to generation, but restricted to air pollutants)	Other external costs of extraction etc. as well as construction of plant etc.

Recovery of metals is excluded since this is not deemed integral to the process, particularly where one is concerned with diversion of biowaste fractions.

It is important to note that none of the air emissions has been explicitly related back to the waste components being combusted. Hence, in Table 26, in which the effect of removing a tonne of compostable waste from an incinerator plant is shown, the nature of the materials being removed is not properly reflected in the externalities being captured. Again, this is shown for the 3% discount rate only. The incinerator is assumed to meet Incineration Directive standards (but no more – in some countries this will lead to overestimates of those external costs which have been quantified).

Plant generating electricity is assumed to do so at 21% efficiency and CHP plants generate energy at 75% efficiency. These are factors which do vary across plant. It is also worth mentioning that across Europe, the proportion of incinerated waste being sent to plant which generates both energy and heat varies across countries.

Different totals are shown exhibiting the potential effects of disamenity and different assumptions regarding the avoided burdens from energy recovery. Unlike the case for landfill, the total is heavily influenced by these two factors. This is due to the fact that:

- It is assumed that incinerators are built in densely populated areas, and that because the calculation is one which runs, effectively, on population density, the effect on the total value of housing stock is significant in this calculation; and
- The energy recovered from incinerators is more significant than for landfill (especially in CHP plant) and it occurs 'at once' so that the effect of discounting is nil. However, an important point here is that to the extent that one is seeking to understand 'what is lost' when one tonne of compostable material is removed from an incinerator, the actual energy which is lost could be relatively small (much smaller than suggested here) if one takes this to be the net energy delivered when one tonne of compostable material is combusted (since the calorific value is low and moisture content is typically high).

The calculation of disamenity is almost certainly unsound (being based essentially on an extrapolation of the landfill case). Setting this aside, it can be seen that the benefits assessment is heavily dependent upon a) what assumption one makes concerning the displacement of burdens from energy generation (including how one accounts for both heat and electrical energy), and b) the degree to which energy is recovered.

As with landfill, by no means all impacts have been valued. Appendix 3 shows also how different rates of energy recovery affect the externality calculus.

Table 26. Effects of Reducing Quantity of Biowaste Incinerated by One Tonne (€ / tonne)

Discount Rate 3%	AU		BE		DE		FI		FR		GE		GR		IR		IT		LU		NL	
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Net Externality (air pollutants, climate change) = A	26.92	38.00	27.94	37.22	24.65	34.05	24.05	33.55	25.98	40.03	25.19	37.65	24.36	36.33	24.36	36.33	25.49	37.25	26.05	34.12	26.10	35.61
Avoided Burdens (Energy Production) Electricity only = B	-4.08	-9.65	-4.08	-8.17	-5.57	-11.14	-2.60	-5.72	-8.91	-13.00	-4.46	-8.54	-2.60	-4.83	-2.60	-4.83	-5.57	-10.03	-1.86	-7.06	-1.86	-7.06
Avoided Burdens (Energy Production) CHP = C	-9.34	-22.07	-9.34	-18.67	-12.73	-25.46	-5.94	-13.07	-20.37	-29.71	-10.19	-19.52	-5.94	-11.03	-5.94	-11.03	-12.73	-22.92	-4.24	-16.13	-4.24	-16.13
Disamenity = D	18.75	93.75	31.25	250.00	21.88	125.00	12.50	75.00	18.75	50.00	31.25	218.75	18.75	43.75	18.75	43.75	12.50	43.75	25.00	93.75	18.75	56.25
Net Externality (excl. disamenity, no displaced burdens from energy generation) = A	26.92	38.00	27.94	37.22	24.65	34.05	24.05	33.55	25.98	40.03	25.19	37.65	24.36	36.33	24.36	36.33	25.49	37.25	26.05	34.12	26.10	35.61
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only) = A+B	22.84	28.34	23.86	29.05	19.08	22.91	21.45	27.83	17.07	27.03	20.73	29.11	21.76	31.50	21.76	31.50	19.92	27.22	24.20	27.06	24.24	28.55
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP) = A+C	17.58	15.93	18.61	18.55	11.92	8.59	18.11	20.47	5.61	10.32	15.01	18.13	18.42	25.30	18.42	25.30	12.76	14.33	21.81	17.99	21.86	19.48
Net Externality (incl. disamenity, no displaced burdens from energy generation) =A+D	45.67	131.75	59.19	287.22	46.53	159.05	36.55	108.55	44.73	90.03	56.44	256.40	43.11	80.08	43.11	80.08	37.99	81.00	51.05	127.87	44.85	91.86
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only) =A+B+D	41.59	122.09	55.11	279.05	40.96	147.91	33.95	102.83	35.82	77.03	51.98	247.86	40.51	75.25	40.51	75.25	32.42	70.97	49.20	120.81	42.99	84.80
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP) = A+C+D	36.33	109.68	49.86	268.55	33.80	133.59	30.61	95.47	24.36	60.32	46.26	236.88	37.17	69.05	37.17	69.05	25.26	58.08	46.81	111.74	40.61	75.73

	PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO			
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High		
Net Externality (air pollutants, climate change) = A	25.31	35.44	25.06	37.06	24.45	33.95	25.70	36.81	25.48	36.72	25.49	37.21	25.51	37.78	25.50	37.40	25.51	37.73	25.48	36.95		
Avoided Burdens (Energy Production) Electricity only = B	-2.97	-7.80	-4.08	-8.17	-2.34	-5.46	-4.08	-8.17	-4.18	-8.38	-4.18	-8.38	-4.18	-8.38	-4.18	-8.38	-4.18	-8.38	-4.18	-8.38		
Avoided Burdens (Energy Production) CHP = C	-6.79	-17.82	-9.34	-18.67	-5.35	-12.48	-9.34	-18.67	-9.56	-19.15	-9.56	-19.15	-9.56	-19.15	-9.56	-19.15	-9.56	-19.15	-9.56	-19.15		
Disamenity = D	9.38	15.63	25.00	156.25	18.75	50.00	31.25	218.75	12.50	43.75	18.75	93.75	12.50	31.25	18.75	93.75	18.75	93.75	12.50	12.50		
Net Externality (excl. disamenity, no displaced burdens from energy generation) = A	25.31	35.44	25.06	37.06	24.45	33.95	25.70	36.81	25.48	36.72	25.49	37.21	25.51	37.78	25.50	37.40	25.51	37.73	25.48	36.95		
Net Externality (excl. disamenity, displaced burdens from energy generation, electricity only) = A+B	22.34	27.64	20.97	28.89	22.11	28.49	21.61	28.64	21.29	28.34	21.31	28.83	21.33	29.40	21.31	29.02	21.32	29.35	21.30	28.57		
Net Externality (excl. disamenity, displaced burdens from energy generation, CHP) = A+C	18.52	17.62	15.72	18.39	19.10	21.47	16.36	18.14	15.92	17.56	15.93	18.06	15.95	18.63	15.94	18.25	15.95	18.58	15.92	17.80		
Net Externality (incl. disamenity, no displaced burdens from energy generation) =A+D	34.69	51.07	50.06	193.31	43.20	83.95	56.95	255.56	37.98	80.47	44.24	130.96	38.01	69.03	44.25	131.15	44.26	131.48	37.98	49.45		
Net Externality (incl. disamenity, displaced burdens from energy generation, electricity only) =A+B+D	31.72	43.27	45.97	185.14	40.86	78.49	52.86	247.39	33.79	72.09	40.06	122.58	33.83	60.65	40.06	122.77	40.07	123.10	33.80	41.07		
Net Externality (incl. disamenity, displaced burdens from energy generation, CHP) = A+C+D	27.90	33.24	40.72	174.64	37.85	71.47	47.61	236.89	28.42	61.31	34.68	111.81	28.45	49.88	34.69	112.00	34.70	112.33	28.42	30.30		

Note: Low and High figures refer to the use of 'sets' of low and high unit damage costs for the various pollutants under consideration. Positive figures indicate environmental benefits of reducing biowaste incinerated by one tonne. CHP = combined heat and power.

4.4 Results for Composting

The analysis for composting incorporates a considerable amount of new material. Table 27 shows which external costs are covered and which are not in this analysis.

Table 27. External Costs of Composting Covered In the Analysis

Class of External Cost / Benefit	Covered	Not Covered
Air Emissions	CO_2, N_20, VOCs Emissions of CO_2 following compost application	Other air emissions (most important of which are ammonia and bioaerosols).
Emissions to Water	It is assumed that excess water is used in the process	
Emissions to Land		Impacts from addition of heavy metals Risks of animal infection through pathogens (thought to be minimal where processes are well-controlled) Full benefits of application of organic matter to land (soil biodiversity etc.)
Other	Energy used on-site	Accidents (relating to e.g. process control) Disamenity (incl. odour etc.) Operator health issues (related to e.g. bioaerosols, ammonia) Plant (construction) related externalities Transport of compost to final market Benefits from use of material for landscaping
Avoided burdens	External benefits from avoided pesticide use (estimate) External costs of manufacture of avoided fertiliser (air emissions only) Reduced phosphogypsum and process wastewater disposal from phosphate fertiliser manufacture Avoided nitrous oxide emissions from nitrate fertiliser application Avoided use of peat (air emissions only)	External costs from manufacture of avoided pesticides Reduction in leaching of nitrate to groundwater Non-air emission external costs of manufacture of fertiliser, extraction of peat.

The avoided burdens, to the extent that they have been quantified, have been related to a ‘typical’ marketing mix for compost products. The analysis is described in more detail in Appendix 4. The results of the benefits analysis are shown in Table 28. Again, these are shown only for the 3% discount rate.

Table 28. External Costs and Benefits of Composting (€ / tonne)

	AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Greenhouse Gases																						
Carbon Dioxide (process) = A	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01
Carbon Dioxide (post application) = B	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.99	-3.14	-2.30	-2.42	-3.13	-3.30	-2.30	-2.42	-3.13	-3.30	-2.30	-2.42	-2.30	-2.42
Methane = C	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48
Nitrous Oxide = D	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16
Other Air Emissions = E	**-0.02**	**-0.08**	**-0.02**	**-0.08**	**-0.02**	**-0.08**	**-0.02**	**-0.07**	**-0.02**	**-0.07**	**-0.02**	**-0.08**	**-0.01**	**-0.05**	-0.02	-0.08	-0.02	-0.07	-0.03	-0.13	-0.02	-0.08
Energy Use = F	-0.62	-1.47	-0.62	-1.25	-0.85	-1.70	-0.40	-0.87	-1.36	-1.98	-0.68	-1.30	-0.40	-0.74	-1.17	-1.67	-0.85	-1.53	-0.28	-1.08	-0.28	-1.08
Fuel Emissions = G	-0.57	-0.95	-0.70	-0.76	-0.28	-0.38	-0.18	-0.23	-0.59	-1.07	-0.57	-0.87	-0.20	-0.53	-0.26	-0.33	-0.36	-0.80	-0.42	-0.49	-0.42	-0.49
Total External Costs = A+B+C+D+E+F+G = H	-12.61	-14.57	-12.74	-14.16	-12.55	-14.24	-11.99	-13.25	-14.05	-15.92	-12.67	-14.32	-12.83	-14.25	-12.85	-14.14	-13.46	-15.34	-12.13	-13.76	-12.12	-13.72
External Benefits from Nutrient Displacement = I	0.13	1.66	0.13	1.66	0.13	1.64	0.12	1.63	0.13	1.73	0.13	1.66	0.13	1.73	0.13	1.64	0.13	1.74	0.13	1.64	0.13	1.64
External Benefits from Pesticide Reduction = J	0.17	0.26	0.59	0.89	0.07	0.11	0.05	0.08	0.24	0.36	0.11	0.17	0.19	0.28	0.70	1.05	0.40	0.60	0.19	0.28	0.58	0.87
External Benefits from avoided nitrous oxide emissions = K	0.03	0.58	0.03	0.58	0.03	0.58	0.03	0.58	0.04	0.62	0.03	0.58	0.04	0.65	0.03	0.58	0.04	0.65	0.03	0.58	0.03	0.58
External Benefits from avoided process wastewater disposal =L	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03
External Benefits from avoided peat extraction = M	0.43	0.57	0.47	0.52	0.33	0.39	0.30	0.34	0.42	0.64	0.43	0.55	0.31	0.43	0.32	0.37	0.36	0.52	0.38	0.43	0.38	0.43
Net Externality = H+I+J+K+L+M = N	-11.84	-11.46	-11.50	-10.49	-11.97	-11.49	-11.47	-10.59	-13.20	-12.54	-11.95	-11.34	-12.16	-11.13	-11.65	-10.47	-12.52	-11.80	-11.39	-10.79	-10.99	-10.16
Net Externality (no CO2) =N - A	-3.28	-2.45	-2.94	-1.47	-3.41	-2.48	-2.91	-1.58	-4.64	-3.52	-3.39	-2.32	-3.60	-2.12	-3.09	-1.46	-3.95	-2.79	-2.83	-1.78	-2.42	-1.15
Memorandum Items																						
Private savings from avoided fertiliser use	0.76	1.91	0.76	1.91	0.76	1.91	0.76	1.91	0.77	1.93	0.76	1.91	0.77	1.93	0.76	1.91	0.77	1.93	0.76	1.91	0.76	1.91
Private savings from avoided pesticide use	0.43	0.43	1.48	1.48	0.19	0.19	0.13	0.13	0.60	0.60	0.28	0.28	0.47	0.47	1.75	1.75	1.00	1.00	0.47	0.47	1.45	1.45

Note: Low and High figures refer to the use of 'sets' of low and high unit damage costs for the various pollutants under consideration. Positive figures indicate external benefits from composting

	PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO			
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High		
Greenhouse Gases																						
Carbon Dioxide (process) = A	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01	-8.56	-9.01		
Carbon Dioxide (post application) = B	-3.13	-3.30	-3.13	-3.30	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-3.13	-3.30	-2.30	-2.42	-3.13	-3.30		
Methane = C	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48	-0.44	-0.48		
Nitrous Oxide = D	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16	-0.09	-0.16		
Other Air Emissions = E	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.02	-0.08	-0.02	-0.08	-0.02	-0.08	-0.01	-0.05	0.00	0.00	0.00	0.00		
Energy Use = F	-0.45	-1.19	-0.62	-1.25	-0.36	-0.83	-0.62	-1.25	-0.64	-1.28	-0.64	-1.28	-0.64	-1.28	-0.64	-1.28	-0.64	-1.28	-0.64	-1.28		
Fuel Emissions = G	-0.40	-0.46	-0.35	-0.72	-0.23	-0.28	-0.41	-0.65	-0.40	-0.60	-0.40	-0.60	-0.40	-0.60	-0.40	-0.60	-0.40	-0.60	-0.40	-0.60		
Total External Costs = A+B+C+D+E+F+G = H	-13.08	-14.59	-13.19	-14.91	-11.98	-13.18	-12.43	-13.96	-12.45	-14.03	-12.45	-14.03	-12.45	-14.03	-13.27	-14.87	-12.43	-13.95	-13.26	-14.82		
External Benefits from Nutrient Displacement = I	0.13	1.72	0.13	1.74	0.12	1.63	0.13	1.65	0.13	1.65	0.13	1.65	0.13	1.65	0.13	1.73	0.13	1.65	0.13	1.73		
External Benefits from Pesticide Reduction = J	0.26	0.39	0.10	0.15	0.05	0.08	0.28	0.41	0.59	0.89	0.59	0.89	0.07	0.11	0.19	0.28	0.58	0.87	0.26	0.39		
External Benefits from avoided nitrous oxide emissions = K	0.04	0.65	0.04	0.65	0.03	0.58	0.03	0.58	0.03	0.58	0.03	0.58	0.03	0.58	0.04	0.65	0.03	0.58	0.04	0.65		
External Benefits from avoided process wastewater disposal =L	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03	0.01	0.03		
External Benefits from avoided peat extraction = M	0.37	0.41	0.35	0.50	0.32	0.36	0.37	0.48	0.37	0.46	0.37	0.46	0.37	0.46	0.37	0.46	0.37	0.46	0.37	0.46		
Net Externality = H+I+J+K+L+M = N	-12.27	-11.39	-12.56	-11.84	-11.45	-10.51	-11.61	-10.80	-11.32	-10.42	-11.32	-10.42	-11.84	-11.20	-12.53	-11.71	-11.31	-10.36	-12.45	-11.56		
Net Externality (no CO2) =N – A	-3.71	-2.38	-4.00	-2.83	-2.89	-1.49	-3.05	-1.79	-2.76	-1.41	-2.76	-1.41	-3.27	-2.19	-3.97	-2.70	-2.75	-1.35	-3.89	-2.55		
Memorandum Items																						
Private savings from avoided fertiliser use	0.77	1.93	0.77	1.93	0.76	1.91	0.76	1.91	0.76	1.91	0.76	1.91	0.76	1.91	0.77	1.93	0.76	1.91	0.77	1.93		
Private savings from avoided pesticide use	0.64	0.64	0.25	0.25	0.13	0.13	0.69	0.69	1.48	1.48	1.48	1.48	0.19	0.19	0.47	0.47	1.45	1.45	0.64	0.64		

Note: Low and High figures refer to the use of 'sets' of low and high unit damage costs for the various pollutants under consideration. Positive figures indicate external benefits from composting

There is no basis for estimating disamenity from compost plants. By far the most important contributor to the external costs of composting (such as they have been valued) are those associated with carbon dioxide emissions. It should be noted that in some studies, the carbon dioxide emissions from the biogenic fractions of waste are omitted. If this were the case in this study, the analysis of costs and benefits would appear to indicate that of all processes, composting is a relatively benign one.

However, this statement has to be conditioned by appreciation of the fact that not all effects have been modelled. So, negative impacts, such as those of bioaerosols, have not been captured, but equally, whilst attempts have been made to capture some of the benefits associated with the application of compost, probably the most significant one, and the one which most closely resonates with 'strong sustainability' arguments – the return of organic matter to the soil – has been captured only 'by proxy'. Another argument in favour of composting may be that the process, being relatively benign, is less likely to give rise to major accidents. This cannot be said of either landfill or incineration.

Some of the benefits from compost utilisation are highly uncertain. A considerable deal of research needs to be undertaken before a clear picture of the costs and benefits of compost application can emerge. At the simple level, this is a process of maintaining / building up soil organic carbon levels. The levels of application which have typically been assumed are not especially high since raising application rates is likely to generate potential negative effects owing to the application of higher quantities of heavy metals to the soil. This illustrates the point that the apparently simple process of applying compost to soil masks a much more complex interaction between soil chemistry and biological activity which is still rather poorly understood (though the state of knowledge clearly increases with time).

4.5 Results for Anaerobic Digestion

Some of the analysis concerning anaerobic digestion (AD) is similar to that carried out for composting. However, emissions are, of course, different, whilst the process also generates energy. Table 29 shows the degree to which external costs and benefits are covered (or not) in this study.

Results for the 3% discount rate case are shown in Table 30 below. The full AD analysis is given in Appendix 5. The comparison with composting shows some reduction in the net external costs relative to composting where one assumes that the energy generated displaces alternative fuels. However, the effects of this are not so marked as with incineration since the energy recovered is typically not so great. It should be borne in mind, however, that the issue of how much energy is generated from incineration of biowastes needs closer analysis than has been possible in this study. A more complex modelling exercise might show that the net energy generated from combusting these wastes would be rather small (see Section 4.3 above). Otherwise, the analysis is similar. There is a suggestion (from the externalities associated with methane and carbon dioxide) that the mass balance in the anaerobic digestion module may need adjustment in line with the assumptions here concerning the methane generated for subsequent combustion and the resultant mass of compost applied to the soil.

Table 29. External Costs of Anaerobic Digestion Covered In the Analysis

Class of External Cost / Benefit	Covered	Not Covered
Air Emissions	CO_2, N_20, CH_4, Dioxins, VOCs, SOx, NOx, Cadmium, Chromium, Lead Emissions of CO_2 following compost application	Other air emissions (most important of which are hydrogen sulphide, HCl, HF, mercury, zinc, PCBs, and bioaerosols). Atmospheric emissions (gaseous and bioaerosols) during post digestion aerobic composting phase.
Emissions to Water	Not quantified. As with incineration, one could take the view that the costs of treating waste waters, being internalised, are an approximation to the external costs of treatment. Strictly, this is not a valid assumption.	
Emissions to Land		Impacts from addition of heavy metals Risks of animal infection through pathogens (thought to be minimal where processes are well-controlled) Full benefits of application of organic matter to land (soil biodiversity etc.)
Other	Energy used on-site (actually netted off against energy generated – see below)	Accidents (relating to e.g. process control) Disamenity (incl. odour etc.) Operator health issues (related to e.g. bioaerosols, hydrogen sulphide, etc.) Plant (construction) related externalities Transport of compost to final market Benefits from use of material for landscaping
Avoided burdens	Air emissions from CHP plant (covering various phases of extraction to generation, but restricted to air pollutants) External benefits from avoided pesticide use (estimate) External costs of manufacture of avoided fertiliser (air emissions only) Reduced phosphogypsum and process wastewater disposal from phosphate fertiliser manufacture Avoided nitrous oxide emissions from nitrate fertiliser application Avoided use of peat (air emissions only)	Other external costs of extraction etc. as well as construction of plant etc. External costs from manufacture of avoided pesticides Reduction in leaching of nitrate to groundwater Non-air emission external costs of manufacture of fertiliser, extraction of peat.

Table 30. External Costs and Benefits of Anaerobic Digestion (€ / tonne)

	AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Greenhouse Gases																						
Carbon Dioxide (process) = A	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30
Carbon Dioxide (post application) = B	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.99	-3.14	-2.30	-2.42	-3.13	-3.30	-2.30	-2.42	-3.13	-3.30	-2.30	-2.42	-2.30	-2.42
Methane = C	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitrous Oxide = D	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71
Other Air Emissions = E	-0.13	-0.22	-0.16	-0.17	-0.06	-0.07	-0.03	-0.03	-0.14	-0.23	-0.13	-0.20	-0.03	-0.11	-0.05	-0.06	-0.08	-0.18	-0.09	-0.10	-0.09	-0.10
Avoided Burdens (Energy Production) Electricity only = F	1.01	5.18	1.01	4.39	1.38	5.98	0.64	3.07	2.20	6.98	1.10	4.59	0.64	2.59	1.89	5.86	1.38	5.38	0.46	3.79	0.46	3.79
Avoided Burdens (Energy Production) CHP = G	1.45	7.44	1.45	6.29	1.97	8.58	0.92	4.40	3.15	10.01	1.58	6.58	0.92	3.72	2.71	8.41	1.97	7.72	0.66	5.43	0.66	5.43
Total external costs, no displaced burdens = A+B+C+D+E = H	-12.21	-13.65	-12.24	-13.60	-12.14	-13.50	-12.11	-13.46	-12.91	-14.38	-12.21	-13.63	-12.95	-14.42	-12.13	-13.49	-12.99	-14.48	-12.17	-13.52	-12.17	-13.52
Total external costs, displaced burdens from electricity = A+B+C+D+E+F = I	-11.20	-8.47	-11.23	-9.21	-10.76	-7.52	-11.47	-10.39	-10.71	-7.41	-11.11	-9.04	-12.30	-11.82	-10.24	-7.63	-11.61	-9.10	-11.71	-9.74	-11.71	-9.74
Total external costs, displaced energy from CHP = A+B+C+D+E+G = J	-10.77	-6.21	-10.80	-7.31	-10.17	-4.92	-11.19	-9.06	-9.76	-4.37	-10.64	-7.05	-12.03	-10.70	-9.42	-5.08	-11.02	-6.76	-11.51	-8.09	-11.51	-8.09
External Benefits from Nutrient Displacement = K	0.11	1.59	0.11	1.59	0.11	1.58	0.11	1.58	0.12	1.65	0.11	1.59	0.12	1.66	0.11	1.58	0.12	1.67	0.11	1.58	0.11	1.58
External Benefits from Pesticide Reduction = L	0.17	0.26	0.59	0.89	0.07	0.11	0.05	0.08	0.24	0.36	0.11	0.17	0.19	0.28	0.70	1.05	0.40	0.60	0.19	0.28	0.58	0.87
External Benefits from avoided nitrous oxide emissions = M	0.03	0.58	0.03	0.58	0.03	0.58	0.03	0.58	0.04	0.62	0.03	0.58	0.04	0.65	0.03	0.58	0.04	0.65	0.03	0.58	0.03	0.58
External Benefits from avoided process wastewater disposal = N	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02
External Benefits from avoided peat extraction = O	0.43	0.57	0.47	0.52	0.33	0.39	0.30	0.34	0.42	0.64	0.43	0.55	0.31	0.43	0.32	0.37	0.36	0.52	0.38	0.43	0.38	0.43
Total external costs, no displaced burdens = H+K+L+M+N+O	-11.46	-10.63	-11.02	-10.00	-11.58	-10.82	-11.61	-10.87	-12.09	-11.10	-11.52	-10.72	-12.29	-11.37	-10.96	-9.88	-12.07	-11.02	-11.45	-10.63	-11.06	-10.04
Total external costs, displaced burdens from electricity = I+K+L+M+N+O	-10.45	-5.44	-10.01	-5.62	-10.21	-4.84	-10.97	-7.80	-9.89	-4.12	-10.42	-6.14	-11.65	-8.77	-9.06	-4.02	-10.70	-5.64	-10.99	-6.84	-10.60	-6.26
Total external costs, displaced energy from CHP = J+K+L+M+N+O	-10.01	-3.191	-9.579	-3.71	-9.611	-2.241	-10.69	-6.464	-8.935	-1.086	-9.944	-4.145	-11.37	-7.648	-8.24	-1.48	-10.1	-3.3	-10.8	-5.19	-10.4	-4.61
Memorandum Items																						
Private savings from avoided fertiliser use	0.572	1.43	0.572	1.43	0.572	1.43	0.572	1.43	0.577	1.442	0.572	1.43	0.578	1.446	0.572	1.43	0.578	1.446	0.572	1.43	0.572	1.43
Private savings from avoided pesticide use	0.43	0.43	1.48	1.48	0.187	0.187	0.127	0.127	0.6	0.6	0.278	0.278	0.472	0.472	1.75	1.75	0.996	0.996	0.469	0.469	1.446	1.446

	PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Greenhouse Gases																				
Carbon Dioxide (process) = A	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30	-9.78	-10.30
Carbon Dioxide (post application) = B	-3.13	-3.30	-3.13	-3.30	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-2.30	-2.42	-3.13	-3.30	-2.30	-2.42	-3.13	-3.30
Methane = C	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nitrous Oxide = D	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71	0.00	-0.71
Other Air Emissions = E	-0.09	-0.09	-0.07	-0.16	-0.04	-0.05	-0.09	-0.14	-0.08	-0.13	-0.08	-0.13	-0.08	-0.13	-0.08	-0.13	-0.08	-0.13	-0.08	-0.13
Avoided Burdens (Energy Production) Electricity only = F	0.73	4.19	1.01	4.39	0.58	2.93	1.01	4.39	1.03	4.50	1.03	4.50	1.03	4.50	1.03	4.50	1.03	4.50	1.03	4.50
Avoided Burdens (Energy Production) CHP = G	1.05	6.01	1.45	6.29	0.83	4.20	1.45	6.29	1.48	6.45	1.48	6.45	1.48	6.45	1.48	6.45	1.48	6.45	1.48	6.45
Total external costs, no displaced burdens = A+B+C+D+E = H	-13.00	-14.40	-12.99	-14.46	-12.13	-13.47	-12.17	-13.57	-12.17	-13.56	-12.17	-13.56	-12.17	-13.56	-13.00	-14.43	-12.17	-13.56	-13.00	-14.43
Total external costs, displaced burdens from electricity = A+B+C+D+E+F = I	-12.27	-10.21	-11.98	-10.08	-11.55	-10.54	-11.16	-9.18	-11.14	-9.06	-11.14	-9.06	-11.14	-9.06	-11.96	-9.93	-11.14	-9.06	-11.96	-9.93
Total external costs, displaced energy from CHP = A+B+C+D+E+G = J	-11.95	-8.39	-11.54	-8.17	-11.30	-9.27	-10.73	-7.27	-10.69	-7.10	-10.69	-7.10	-10.69	-7.10	-11.52	-7.98	-10.69	-7.10	-11.52	-7.97
External Benefits from Nutrient Displacement = K	0.12	1.66	0.12	1.67	0.11	1.58	0.11	1.59	0.11	1.58	0.11	1.58	0.11	1.58	0.12	1.67	0.11	1.58	0.12	1.67
External Benefits from Pesticide Reduction = L	0.26	0.39	0.10	0.15	0.05	0.08	0.28	0.41	0.59	0.89	0.59	0.89	0.07	0.11	0.19	0.28	0.58	0.87	0.26	0.39
External Benefits from avoided nitrous oxide emissions = M	0.04	0.65	0.04	0.65	0.03	0.58	0.03	0.58	0.03	0.58	0.03	0.58	0.03	0.58	0.04	0.65	0.03	0.58	0.04	0.65
External Benefits from avoided process wastewater disposal = N	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02	0.01	0.02
External Benefits from avoided peat extraction = O	0.37	0.41	0.35	0.50	0.32	0.36	0.37	0.48	0.37	0.46	0.37	0.46	0.37	0.46	0.37	0.46	0.37	0.46	0.37	0.46
Total external costs, no displaced burdens = H+K+L+M+N+O	-12.21	-11.27	-12.38	-11.48	-11.61	-10.87	-11.37	-10.48	-11.05	-10.02	-11.05	-10.02	-11.57	-10.80	-12.28	-11.35	-11.07	-10.04	-12.21	-11.25
Total external costs, displaced burdens from electricity = I+K+L+M+N+O	-11.48	-7.08	-11.37	-7.09	-11.03	-7.93	-10.36	-6.10	-10.02	-5.52	-10.02	-5.52	-10.54	-6.30	-11.24	-6.85	-10.03	-5.54	-11.18	-6.75
Total external costs, displaced energy from CHP = J+K+L+M+N+O	-11.16	-5.263	-10.93	-5.185	-10.78	-6.661	-9.926	-4.19	-9.575	-3.566	-9.575	-3.566	-10.09	-4.341	-10.8	-4.89	-9.59	-3.59	-10.7	-4.79
Memorandum Items																				
Private savings from avoided fertiliser use	0.578	1.446	0.578	1.446	0.572	1.43	0.572	1.43	0.572	1.43	0.572	1.43	0.572	1.43	0.578	1.446	0.572	1.43	0.578	1.446
Private savings from avoided pesticide use	0.642	0.642	0.246	0.246	0.125	0.125	0.689	0.689	1.48	1.48	1.48	1.48	0.187	0.187	0.472	0.472	1.446	1.446	0.642	0.642

Note: Low and High figures refer to the use of 'sets' of low and high unit damage costs for the various pollutants under consideration. Positive figures indicate external benefits from composting.

This argument is given greater force when one considers that very few datasets for anaerobic digestion report the emissions to the atmosphere during the post-digestion aerobic composting phase (where this takes place). Not only would one expect further methane to be emitted from the residual anaerobic processes, but carbon dioxide would also be emitted from the aerobic process. As such, the resultant mass of compost applied to land might be expected to be somewhat less than has been assumed here. That having been said, this affects the attribution of benefits which are, by and large, relatively small (so reducing the quantity of biomass assumed to be applied to land to say 300kg per tonne of biowaste would not affect the results appreciably).

County-to-country variation mostly reflects the assumptions made concerning external costs of energy being displaced. Where this assumption is implied, and where the displaced source has a relatively high external cost, the net externality is lowest.

4.6 Note on Transport Externalities

In this study, the principal issue of interest is the effect of the switch from one waste management system to another. As such, one is interested in the changes in transport externalities incurred by switching waste management systems.

In a number of studies concerning the environmental impacts of waste management, the approach has been to specify transport routes and modes that are inextricably linked to one or other waste management system (e.g., Coppers & Lybrand et al 1997; Broome et al 2000; CSERGE et al 1993; AEA 2001). The rather obvious point to be made is that these distances vary enormously from one situation to another, so much so that the distances themselves may be important in determining the treatment options pursued.

In studies which have looked at the transport costs incurred in the collection of organic wastes, some studies have assumed that the bulk of waste is delivered to civic amenity sites. This was the assumption used by Coopers and Lybrand and CSERGE (1997) in an earlier study for the Commission. Furthermore, such studies have assumed (implicitly or explicitly) that the journey is made specifically for that purpose. This effectively denies the potential of the very changes being examined. This is not to say that such materials might not continue to be delivered to Civic Amenity sites / containerparks / recycling centres. However, to the extent that such materials are not always composted in all countries, it is quite clear that similar transport externalities should be attributed to the landfilling / incineration of materials collected in the same way (on a proportionate basis) to reflect the role of Civic Amenity sites (and other similar facilities) in the collection of waste materials.

4.6.1 External Costs of Separate Collection of Garden Wastes

In this study, the delivery of garden wastes to Civic Amenity sites / containerparks to some extent constitutes the baseline in a ‘no separate collection’ scenario. Of course, many countries, among them Germany, Austria, Flanders (Belgium), Denmark and the Netherlands are already collecting a significant proportion (estimates as high as 90% in Denmark) of all garden wastes through this route for composting. But in those countries where such material is not currently composted, the net transport externalities incurred by the decision to collect such materials separately for composting are zero, at least in terms of the collection side (since the materials are being delivered there anyway – all that is changing is that those making deliveries are being given the opportunity to put materials in containers specifically for garden waste as opposed to ones for mixed waste. Furthermore, disposal costs are being avoided).

For those materials which are separately collected at Civic Amenity sites / containerparks / recycling centres where they were not previously, the net effect on transport externalities arises through consideration of the change in the external costs of the delivery route to the treatment facility. This reduces to a question as to whether a compost facility is likely to be closer to, or further away from the Civic Amenity site, than the 'counterfactual' treatment facility.[21]

One could manufacture arguments as to why the distance is likely to be greater or smaller, but the fact of the matter is that no one can say without examination of the specific situation. The most plausible argument might be that since capacities of compost plants are typically lower than for mass-burn incinerators and landfills, the more 'decentralised' nature of the compost facilities might be more likely to reduce the transport externalities than to increase them. On the other hand, the potential for smaller thermal treatment plants to penetrate the market (which is not known at present) might alter this perspective. Furthermore, mass-burn facilities can deal with all wastes, whilst compost plants deal with specific fractions.

The matter is further complicated by the issue of residuals. All plant other than landfills will tend to generate residual materials which require treatment off-site. For compost, this will be rejected materials which might typically be sent to landfill. For incinerators, the fly-ash will require disposal to hazardous waste landfill with bottom ash potentially used in construction. Again, the distances involved are amenable only to guesswork, and in some countries, they may change with the end to co-disposal required by the Landfill Directive. It makes little sense to make such guesses here.

The upshot of this discussion is that at the least for the garden waste fraction, the transport implications of separate collection are not knowable, but likely to be close to zero. It cannot be stated, unequivocally, whether the impact will be positive or negative. Positive situations in some specific circumstances are likely to be offset by negative ones elsewhere. The picture is also a dynamic one. The more the activity proceeds, and the more the market for materials is expanded, the greater the density of the network of compost facilities. This would incline one toward a view that transport externalities could be reduced.

4.6.2 Kerbside Collections

The effect on transport externalities of kerbside collections is also not straightforward to predict. Intuitively, the response might be to suggest that the transport externalities

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[21] Note, in the case of garden waste, it is more likely to be a compost facility rather than a digestion facility owing to the lignin content of the feedstock.

must increase. Yet many countries are already collecting biowaste in such a way as they reduce the frequency of residual waste collection. The effect is most pronounced in Southern Member States. In parts of Italy, flat-bottomed trucks with no compaction mechanism collect kitchen waste fractions three times weekly. This has enabled the frequency of residual waste collection to fall from three times weekly to once weekly. Although the capacities of the trucks are smaller than for refuse collection vehicles (RCVs), and so the external cost per unit weight of material collected might be assumed to be greater, the fact that some of these vehicle fleets are powered by electricity makes it possible that the net external costs of the system are no greater than (and possibly less than) those of the system they effectively displace, in which RCVs collect material three times weekly. Note that where kitchen waste only is collected, the absence of need for compaction mechanisms also implies an energy saving.

Similar effects can be observed in Northern Member States. Here, it is increasingly common to see collection of residual / rest waste operating on alternate weeks to the collection of kitchen and garden waste. Sometimes, the collection of the biowaste fractions increases in frequency in the summer months.

Note that most systems operating in Northern Member States collect both kitchen and garden wastes. This necessitates use of compactors to achieve high bulk densities. From the perspective of transport externalities, there are interesting, inter-related effects. In the first instance, the decision to collect garden waste as well as kitchen waste typically results in higher per capita waste collection. It is not always clear whether this increase occurs at the expense of home composting (or leaving grass cuttings on a lawn) or whether it results in less garden waste delivered to civic amenity sites. To the extent that it does the former, the transport externalities are increased. But, to the extent that it also does the latter, transport externalities may be reduced depending upon the extent to which journeys to the Civic Amenity site made by the householder would have been made anyway.

Similar discussions in respect of distances to treatment plant apply as with garden waste systems (see above). It is not possible to estimate a net effect on transport externalities associated with the changing treatment systems.

Again, the net effect depends upon a range of factors. However, under the assumption that biowaste collection can, because it removes fermentable fractions from rest / residual waste, reduce the frequency of rest / residual waste collection, the net effect is again likely to be close to zero. This, incidentally, is also reflected in the collection costs in best practice schemes. In the next Chapter, it is argued that these are likely to be close to zero.

4.6.3 Transport Externalities and Degrees of Internalisation

Quite apart from the points made above, where road transport is used to move materials form one location to another, it may well be that transport externalities are already internalised in the costs of road transport through fiscal measures such as fuel taxes. Consequently, there may be cases where double-counting exists in the calculation of transport externalities associated with waste collection since these may already be internalised in the cost of fuel. Such taxes are increasingly common in Europe.

There is, at present, a debate as to whether, and if so, which of these taxes, is ‘environmental.’ That aside, the effect in the real world is the same (to make transport more expensive). Appendix 6 gives a slightly rough-and-ready analysis of the degree to which transport externalities are likely to be internalised in collection costs through existing levels of duties and taxes on diesel fuels in different countries. This suggests that even if one does not accept the argument put forward here (that the net change in externality is difficult to estimate), the external costs of waste transport may already be internalised to a significant degree by duties paid on fuel in different Member States.

4.7 Note on Mechanical Biological Treatment

In the wake of growing interest in mechanical biological treatment (MBT), a number of studies have been produced looking at this approach to managing waste. None thus far has attempted to assess the external costs of the treatment. However, there is a growing body of environmental data available for such an analysis.

Recent studies include those undertaken by the Umweltbundesamt (Lahl et al 2000), the VITO (2001) study in Flanders and the pan-European study by Zeschmar- Lahl et al (2000). The first and last of these provide the most wide-ranging analysis of the processes. These processes are quite diverse. The diversity reflects the different objectives (and hence designs) of the plants. This makes it somewhat awkward to describe a ‘typical’ process. Furthermore, the ultimate destiny of the different fractions makes it difficult to give a clear picture of emissions and environmental impacts other than in the case of specific plants.

Given that the shifts to MBT are not central to the interests of this study, no attempt has been made to elicit external costs of such processes. However, it should be noted that where a municipality’s strategy is pushed further towards source separation of municipal waste by virtue of a specific policy instrument, the impact on that municipality’s treatment of residual waste may include a shift towards mechanical biological treatment approaches in conjunction with ‘smarter’ use of high-calorific fractions in preference to treatments such as mass burn incineration, which may be made less likely by a requirement for source separation of municipal waste.

There are three reasons for positing this: a) the suitable scale of thermal treatment plant decreases (introducing diseconomies of reduced scale into the equation where incineration is concerned); b) under some situations, the calorific value of residual material may become sufficiently high as to cause problems for conventional grate incinerators; and c) in the face of local opposition to incineration (where this occurs), the requirement to source separate biowastes makes the achievement of Landfill Directive targets through a combination of source separation / MBT more viable. This would be especially true if stabilised waste meeting certain conditions was treated as though it was no longer ‘biodegradable municipal waste’ for the purposes of the Article 5 targets for Member States under the Landfill Directive (and the rationale for this lies in the intent to stabilise the biodegradable fraction of residual waste).

Already, the process treats something of the order 4% of residual waste in Germany (Zeschmarr-Lahl et al 2000). In excess of 30 plants are already in existence. In Austria, MBT is an accepted method of pre-treating waste before landfill. Waste which has not undergone an acceptable pre-treatment 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 and the process meets 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). At least eleven plants already exist and the most recent statistics suggest that 6.3% of waste was being treated in this way. As a percentage of residual waste, this figure equates to 12.7% of the total whilst 57.6% of residual waste is still landfilled. This suggests that a major expansion in the use of MBT is likely in the face of existing regulations.

Many plants exist in Italy and new plants are emerging in the Netherlands. The Italian ‘Ex-Maserati’ plant on the outskirts of Milan Town is one of the largest in the world. Flanders, also, looks likely to develop MBT capacity significantly in conjunction with thermal treatment facilities as mass-burn incinerators have become less popular. Plans are for four new MBT facilities.

5.0 THE ECONOMIC ANALYSIS OF OPTIONS FOR MANAGING BIODEGRADABLE MUNICIPAL WASTE – FINANCIAL COSTS

When discussing the financial costs of waste management options, the tendency has been to report costs on a gate fee basis. Gate fees are, effectively, unit prices paid by the customer for the service provided (e.g. the price paid for the landfilling, or incineration of a tonne of waste). From the perspective of a local authority seeking to understand the annual costs of managing its waste, this makes some sense. However, more generally, gate fees do not necessarily represent the financial resources required to establish and operate a waste treatment facility. Furthermore, depending upon contractual design, they may fluctuate over time.

The degree to which reported gate fees and the 'true' financial costs of waste management diverge is likely to be related to the development of the market for waste treatment facilities and the structures of ownership and responsibility within that market. Changes in legislation can significantly affect the amount of waste sent to one or other treatment facility, and the consequences of such changes are that in market situations, gate fees within the market place have to take account of, or to anticipate, these changes. On the other hand, where systems are in public hands, the adjustment is less likely to occur through the market, and will be controlled by the relevant actors.

This Section addresses the costs of those technologies of greatest concern to this study. The following processes / technologies are considered:

- Separate collection of compostable wastes;
- Landfilling;
- Incineration with energy recovery;
- Composting;
- Anaerobic digestion; and
- Mechanical biological treatment (MBT).

In the case of the typical treatments, it has been necessary to resort to the use of typical gate fees in this study. Gate fees still vary enormously across Europe despite attempts to harmonise waste legislation. This partly reflects continuing differences in legislation and standards, but it also reflects the way in which markets for the supply of waste management services varies across countries. The degree to which the gate fee approach represents a problem is discussed in passing and is raised as a more specific concern in Chapter 6. In the general case, it should not be assumed that gate fees will be representative of underlying costs. This is especially true in the context of major regulatory changes such as the Landfill Directive, the implementation of which will inevitably lead to strategic behaviour and significant changes in the market for waste management services.

In this chapter, the factors which affect underlying costs are discussed. However, what is being reported are gate fees. It is important to understand the distinction between the two. In an analysis such as this, it would be preferable to identify costs. The gate fees charged do not, in all cases, change in step with changes in costs.

5.1 Costs of Separate Collection

In the current context, one of the most important issues is the cost incurred by the implementation of source separation of biowastes. If local authorities are required to collect compostable materials separately, how will this affect costs?

The collection of resources from the waste stream and the collection of residual fractions is undertaken in different ways in different countries. There is no one 'right' way to organise collection systems, though there is good reason to believe that some are more effective than others. Where systems work well, they tend to be evolutionary, and incorporate the outcome of what is essentially a learning process. It is also vitally important, when considering separate collection of compostables, to understand the implications for existing collection schemes.

It is not straightforward to estimate the costs of implementing separate collection schemes where they do not already exist. The normal presumption is that the costs of waste collection will increase significantly as a consequence of the introduction of separate collection systems. However, this need not be the case for compostable wastes.

The separate collection of biowastes has consequences for the collection system as a whole. Successful collection at kerbside / doorstep of the kitchen waste fraction can facilitate a reduction in the required frequency of collection of the residual waste fraction. This already happens in various municipalities in a number of countries, and is an especially important consideration in Southern countries where the climate demands more frequent collection of putrescible wastes (though this 'frequency reduction' effect is by no means confined to Southern Member States).

The other interesting point concerning kitchen wastes is that they are dense. For this reason, there is not the same requirement for compaction vehicles to achieve higher bulk densities that there might be with, for example, packaging materials, or mixed kitchen and garden wastes, or residual municipal waste. There is, therefore, no reason why one has to incur the expense of such vehicles. Essentially, flat-bottomed trucks can be used to collect the material, and again, this is already the case in municipalities in some European countries (with commercial and industrial wastes as well as municipal waste).

Where separate collection of only kitchen wastes is in place, collection of garden wastes could be achieved through two possible routes (or a combination of these), though it makes sense to encourage home composting of this material as far as possible:

1. through requiring householders to take such waste to civic amenity sites (as happens, for example, in Denmark, partly because municipalities wish to reduce costs to themselves); or

2. through implementing periodic collections for bulk garden wastes (perhaps limited to spring / summer months).

These decisions should also be considered in the context of the effects on waste minimisation through home composting. Schemes which collect both kitchen and garden waste at the kerbside / doorstep, as well as being potentially more costly to run because of the need to make use of compactors, are likely to draw much more material from the waste stream (and so be more expensive to run both on a per tonne basis, and on a per household / per inhabitant basis). Unless such schemes are also accompanied by variable charging (i.e. pay-as-you-throw) schemes, there will be little incentive for householders to engage in home composting.

Appreciating these possibilities, it is easier to see why it may not be true to say that separate collection schemes for organic wastes incur considerable additional costs. Collection methods which are optimised for their 'systemic' suitability to a specific area can ensure that any additional costs are kept to a minimum, and intelligent adaptation of schemes may even lead to savings in collection costs relative to those which prevailed before separate collection was introduced.

Even where savings are not realised in collection per se, they may be realised through the avoided costs of disposal / treatment of the residual waste fraction. In a number of countries, the costs of composting are already below those of alternative treatments (which have exhibited a tendency to increase through use of economic instruments and regulations). For example, in Austria, one source suggests that collection and recycling of separate collected biogenic waste costs €30 per tonne or so less than collection and treatment of residual waste.[22]

Another way of minimising additional costs (relative to the situation in which no such collection scheme exists) is to use split-bodied vehicles. These have certain problems but the potential exists to use these to collect kitchen waste and residual waste fractions. Given that the incremental costs of such vehicles above and beyond standard refuse collection vehicles may be of the order €15,000, and given that these may collect of the order 2,000 tonnes per annum, the annualised incremental costs may be relatively small, though much depends upon the material capture rates in such schemes. At a capture rate of 50%, the additional cost of the collection system might be €6-7/tonne collected, though this would fall at higher capture rates (which, as Chapter 3 showed, are already being achieved nationwide in some countries).

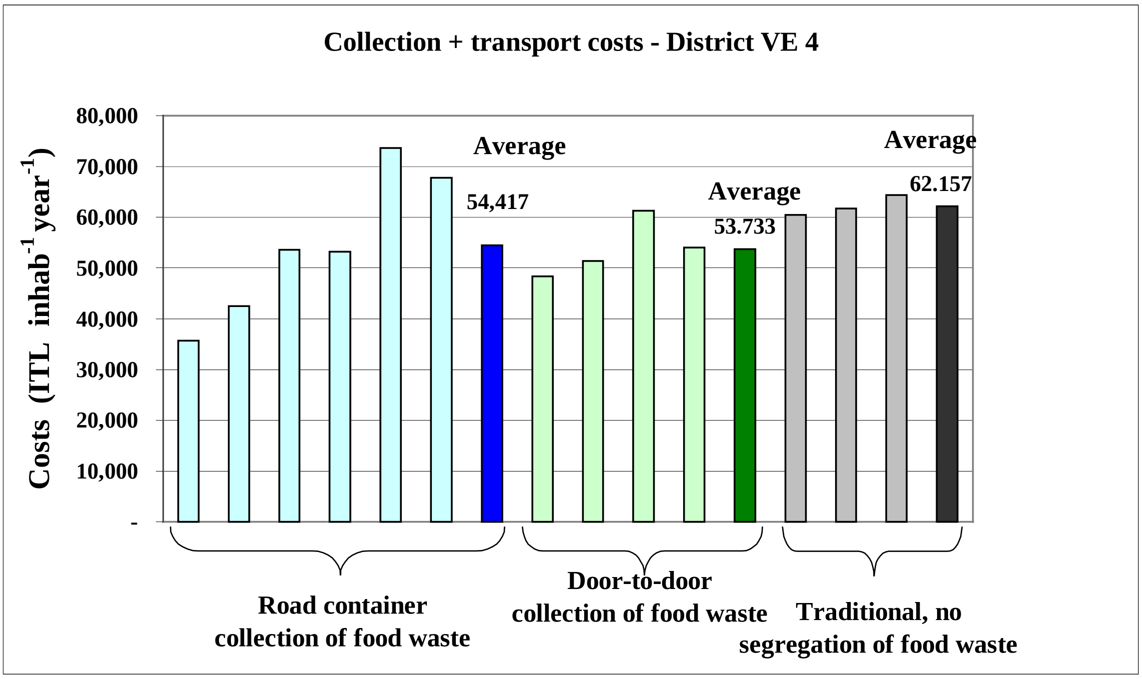
Note that comparisons of collection costs per tonne of separately collected kitchen waste and of residual waste become relatively meaningless once one appreciates the significance of considering collection approaches as systems. What happens in many schemes is that the collection of residual waste becomes less expensive (measured in per tonne terms) as a result of reduced collection frequencies. At the same time, the

[22] Personal comm.. Wolfgang Stark, GUA Group.

lower capture (per household pass) of kitchen waste collection may make the material more expensive to collect on a per tonne basis, but the 'system' collection costs remain more or less constant.

In Figure 12 below, the effects of different approaches to collection in a single district of Italy are illustrated. The system collecting kitchen waste at kerbside guarantees higher purity of compostable waste collection, but it is also the cheapest system to operate on average. Similar findings are now observable in Catalonia in Spain.

Figure 12: Cost Comparison (ITL.Inhab^{-1}.Year^{-1}) For Different Collection Schemes In A Single District



Source: Favoino (2001)

In the United Kingdom, as elsewhere, those municipalities which have achieved high rates of recycling and composting all include collection of biodegradable waste in their collection systems. The approach is frequently based upon collection of both kitchen and garden waste on alternate weeks. The municipality of Daventry, for example, achieves rates of recycling and composting of the order 40% at a net cost of approximately €10 per household relative to the standard (UK) collection system (including dry recyclables). Arguably, in this type of collection system, this could fall if the collection system was targeted at kitchen wastes and the vehicle stock was changed accordingly.

In Sweden, collection of organic wastes may occur once every two weeks, and in some systems, the residual fraction is collected only once every month. This again has the effect of keeping increases in waste collection costs at a low level.

In Denmark, consideration of the establishment of separate collections for organic household waste (as opposed to garden waste, which is already effectively collected in Denmark) are underway. *Waste Management Plan 1998-2004* states that:

Establishment of separate collection and biological treatment of organic fractions requires that investments are made in collection systems and treatment capacity. Calculations show that costs of total operation (including interest payment and depreciation) will not necessarily exceed costs of today's single-string system (Danish EPA 1999).

Having made these points, it is worth adding that it is certainly not necessarily true that in all situations *at present*, the separate collection of compostable materials leads to zero increase in waste management costs to householders. It does seem likely, however, that where waste management systems are carefully designed, increases in collection costs are more likely to be attributable to separate collection of dry recyclables, which are typically of much lower bulk density, and which have little or no impact upon the required frequency of residual waste collection where no biowaste collection is in place. Equally, as with composting, the costs of recycling net of revenues generated may be comparable, or lower in cost, than residual waste collection and alternative treatments.

For the purposes of this study, it is assumed that for each tonne of biowaste collected separately, the incremental cost relative to existing collection is between €0-15 per tonne of biowaste collected (the incremental costs of introducing separate collection are greater when expressed ‘per tonne of biowaste collected’ than when expressed ‘per tonne of total waste collected’, or ‘per household served’). This is based on the view that even though collection costs per tonne of collected biowaste may be higher than for residual waste, offsetting effects (shift to alternate / less frequent collection of residual waste) will act to restrain any increase in costs. The possibility that net costs of collection fall is a very real one, but this assumption has not been used in the modelling.

Note that some will argue that collection costs should reflect the population density of the location from which waste is being collected. That is likely to be true, but here one is interested in a combination of:

- the incremental costs over and above a situation in which no such scheme exists - the costs of the existing collection scheme will vary by type of location; and /or
- the incremental costs (potentially negative) of making participation in existing schemes mandatory where it might not be today.

Recognising these facts, the additional costs for the total collection system are likely to be low where the system is well-designed, and especially where legislation makes separate collection mandatory for householders, or where variable charging encourages householders, through differential collection costs, to separate out biowastes (and dry recyclables).[23] These are the sorts of implementing measures that might flow from any policy which requires municipalities to collect biowastes separately, and indeed, in areas of Europe where such collection is widespread, these measures are already in place.

5.2 Results for Landfill

The financial costs of landfilling include the cost of site acquisition and the engineering costs for landfilling, the latter including any installations put in place to collect and generate energy from landfill gas. Acquisition costs may reflect the age of the site concerned. Engineering costs may depend upon the situation of the landfill with respect to major aquifers. In addition, operating costs include monitoring and leachate treatment, as well as ongoing engineering costs (since modern practices require ongoing activity in respect of capping etc.).

One earlier study gave figures on the basis of costs in one country with the figures extrapolated to other countries on the basis of relative costs of capital, labour and other factors. Whilst intuitively plausible, the key point is that the relative costs in different countries are affected by the type of landfill under consideration and the existing regulatory regime. Since one is interested here in municipal waste, one clearly needs to be looking at municipal waste landfills, but the costs of running these will be determined by the extent to which other wastes are accepted, and the regulatory / licensing regimes in different countries. Furthermore, geological characteristics will affect engineering costs, and the size of the landfill will determine the contribution that these make to unit prices charged for landfill disposal.

Probably more significantly, the degree to which Member States implement policies which reflect the waste management will influence the quantities of waste being landfilled. It is the rate of tipping, as much as size per se, that is likely to affect the unit costs. Where waste is being diverted from landfill in significant quantities, the tipping rates at sites are likely to be lower, requiring gate fees to be higher if the costs of fixed investments are to be covered. It is often felt that diversion of waste from landfill is greatest in countries where landfill gate fees are higher. It may be equally true, however, to state that where landfill gate fees are higher, they are higher because waste is being diverted from landfill (tipping rates are lower). Energy recovery is likely to make some positive contribution to revenues, though the magnitude of revenues is clearly influenced by energy policies in the country concerned.

Taken together, these suggest that in-country landfill costs will exhibit some variation, but also, that the regulatory regime will be a key determinant of costs (both operational, and engineering). There is variation both within and across countries, only part of this being related to differences in the cost of input factors. Currently, the

[23] Note that increasing attention is being paid to designing collection systems and incentive mechanisms which encourage (or which do not discourage) home composting. This helps to minimise the degree to which convenient biowaste collections simply increase quantities collected by pulling into the collection stream material which could readily be composted at home.

cross-country variation in costs is enormous given the different approach to regulation of landfilling across the Member States and Accession Countries.

It is important to emphasise that costs and gate fees are different. To the extent that the landfill is owned by a municipality, arguably the distinction is not relevant since the costs and the price could be considered to be the same, although even here, accounting conventions may affect the costs as reported. Where the landfill is not owned by the municipality, the fee charged for landfilling municipal waste and the costs of supplying the service are likely to be different. The reasons for the differences and their magnitude, as well as their direction, are likely to be influenced by the degree to which it can be said that there is a ‘market’ for waste management services (spanning landfill and other treatments), as well as the strategies of the companies concerned. The latter are likely to be affected by the Member States’ implementation of the Landfill Directive, making the next few years, potentially, a period of considerable uncertainty in the price charged for provision of landfill services.

Landfill operators’ strategies also vary. Some may seek to maximise revenue in the short term, others may seek to preserve void space in the hope of benefiting from higher gate fees later in time. From this perspective, the ratio of non-recurring capital costs to ‘recurring capital costs plus operating costs’ becomes an important ratio, again being determined by regulatory requirements.

Arguably, with the passing of time, the underlying costs for landfilling are likely to converge towards the higher end of existing ranges, though the influence of the Landfill Directive on gate fees charged may also be considerable, especially in countries where co-disposal (of hazardous and non-hazardous wastes) is currently practiced (not least because the higher gate fees from hazardous materials may be lost, requiring costs and profits to be covered by non-hazardous waste gate fees).

In Table 31 below, the *gate fees* for landfilling as suggested by the survey carried out for this study are shown. These are compared with the ‘costs’ quoted in Coopers & Lybrand et al (1997). It would be preferable to use cost data but this is not available. The Table, which includes some gaps, highlights the likely limitations of using an approach which does not account for the differing approaches which exist in Member States and Accession States. Notwithstanding the points made about the potential divergence between ‘gate fees’ and costs, the Table suggests greater variation than could be explained by an approach which assumes that costs vary only with the relative cost of certain input factors.

Also noteworthy from the Table is the significance of the landfill levies now in place. These are a significant component of overall costs of landfilling in those countries where they are in force. Account of the existence of these taxes should be made for in the cost-benefit analysis.

In the context of cost-benefit analyses concerning what may happen in the future, using current landfill gate fees would give a poor representation of what the relative costs and benefits might be of different options. Landfill costs, and probably gate fees, are likely to rise in those countries where gate fees are currently low. The extent of any such rise will depend upon, amongst other things, the way in which different Member States implement various aspects of the Landfill Directive, including the pre-treatment requirements. For example, in work under preparation (Eunomia et al, 2002), the costs of landfilling in Italy might be expected to rise to €50 from what were recently low levels (€20 or so).

Table 31. Costs Of Landfilling In Different Countries, (€ / tonne MSW) (note becoming irrelevant for DK and NL)

LANDFILL	**Coopers & Lybrand et al (1997)**				**This Study**				
	No Energy Recovery		With Energy Recovery		Landfill (excl. levies)			Landfill Levy	
	Urban	Rural	Urban	Rural	Low	Med	High	Tax (best practice / low rate)	Tax (not best practice / high rate)
Austria					55		110	43	
Belgium	22	17	22	17					
Flanders					52		56	Flanders – €52 with energy recovery, €55 without	
Wallonia					41	0	42	25	43
Denmark	48	28	47	27		44		50	
Finland					16	28	56	15	
France	21	15	20	14	31	46	85	9	
Germany	51	29	51	28	35	120	220	None	
Greece	20	15	20	14	5		21	None	
Ireland	34	23	34	22	34	44	78	22	
Italy	25	18	24	16	50		70	10 Note – set at regional level	50 Note – set at regional level
Luxembourg	53	37	52	36	123		147	None	
Netherlands	36	21	36	21		75		60	
Portugal	21	15	20	15	6		15	None	
Spain	25	14	24	16	9	15	30	None	
Sweden					22		82	30	
UK	26	19	25	18	10	24	34	19	
Cyprus					No data (estimated as Greece)			None	
Czech Republic					15		25	1	
Estonia					No data (estimated as CZ Rep)			Landfill tax set by local authorities	
Hungary					No data (estimated as CZ Rep)			None	
Poland					No data (estimated as CZ Rep)			None	
Slovenia						5		In preparation	

5.3 Results for Incineration

Intuitively, the costs of incineration with energy recovery might be expected to vary somewhat less than the costs of landfilling. The capital cost element of incineration in the total cost structure is somewhat greater than for most other treatments. Consequently, the source of variation in costs is more likely to be differences in the cost of capital, and scale (since there are considerable economies of scale for incinerators). To the extent that differences in the cost of capital may be being reduced (with completion of the Single Market and the adoption of the Euro in many Member States), so a substantial proportion of costs for an incinerator of a given capacity is likely to be relatively invariant across countries. A key issue then becomes the relative scale of operation. Indeed, as source separation proceeds, the capacity requirement falls to levels which are likely to make alternative thermal treatments, such as gasification and pyrolysis, competitive on cost terms (where they are not already).

On the other hand, different countries have operated with different emissions standards. Furthermore, different plants have been more and less successful in meeting these. As with landfill, therefore, there is likely to be some variation in costs associated with the degree to which emissions are regulated. The tighter are standards, the greater the requirement for investment in flue gas cleaning equipment, and the greater is the requirement for additional capital investment. The newly agreed Incineration Directive is likely to lead to some harmonisation here. Those operating at lower standards are likely to have to make greater efforts to change than those with higher standards already in place (some of whom may have to make minor, if any, changes).

Energy recovery is an important aspect of incinerator plant operation. Sales of both electricity and thermal energy can be made, though in practice, whilst many incinerators generate electricity, not all are attached to district heating schemes. Those that are, such as the majority in Denmark and Sweden, may be able to benefit from sales of both heat energy and electricity.

Energy purchases are subsidised in some countries on the basis that energy from waste is considered 'renewable', a strange concept given the widespread agreement that the best way of dealing with waste is to minimise its production. Most notably, the non-fossil fuel obligation (NFFO) in the UK (no longer in place) and the Italian CIP schemes have led to support for energy production from incineration. In Italy, subsidies for energy produced during the first 8 years (CIP 6/92), set at 0.125 Euro/KWh, reduce incineration gate fees by around € 60 Euro per tonne of waste. The UK subsidy via NFFO was rather less. The subsidy has been between €0.03 and €0.04 per kWh (based on ESD (2000)). Assuming electricity generation of the order 500kWh per tonne of municipal waste, the subsidy amounts to €14- €20 per tonne of waste. In Flanders, if electricity is produced by treating waste (or another alternative way of electricity production), the producers receive 0.075 Euro per kWh which is supplied to the electricity network.

In the UK, incinerators can also issue packaging recovery notes (or PRNs) against 19% of municipal waste incinerated in recognition of their role in recovering packaging waste. These can be sold to companies obligated under the Producer Responsibility

(Packaging Waste) Obligations as evidence of compliance with their recovery targets (which were established to ensure compliance with the Packaging Directive). The 'recovery PRN' price in 2000 was of the order €13-€16, implying a subsidy of €2.6-€3.2 per tonne of MSW. The level of revenue generated is likely to increase in 2001 as the market for PRNs tightens (as company obligations to recycle / recover packaging increase in the UK).

Ironically, whilst these subsidies exist, a number of countries are now taxing incineration as well as landfill. Flanders and Denmark (as well as Norway) already do this. In addition, taxes on NOx emissions affect incineration in some countries, notably Sweden, where although the NOx tax is refunded to energy producers, it is refunded in proportion to thermal output. Incineration, being a relatively large producer of NOx per unit of thermal output in Sweden, is a net loser under this refunding mechanism (see ECOTEC et al 2001, Hoglund 1999a;1999b).

Also important is the approach taken to dealing with residues. In several countries, bottom ash is used in construction applications. In some, fly ash is also used. Some concerns remain regarding the use of these as, especially in the case of fly ash, the weathering of the material and the eventual dismantling of the construction itself may lead to new concerns regarding pollution.

Although other applications exist, ash is still landfilled in many cases. The Landfill Directive may affect the costs of disposal to landfill, especially where co-disposal currently occurs, depending upon how pre-treatment is defined, and whether ashes, and if so, which ashes, are defined as hazardous waste. The costs of treating and disposing / recycling ash residues varies considerably across countries and will continue to do so. Table 32 below again gives a comparison of the figures obtained for this study against the Coopers and Lybrand et al (1997) report.

Note that, reflecting the discussion above, with the exception of data from Sweden (supplied by RVF), and the UK (which includes subsidies for energy generation and the issuing of PRNs), the costs vary less across countries. The Swedish figures are low reflecting the sales of heat energy. UK figures are also low reflecting the range of implicit and explicit subsidies to incineration. Some local authorities in the UK have the gate fee effectively lowered even further by a system of financing of capital projects in which qualifying projects have the capital sum effectively paid for by central government (the credits scheme under so-called Private Finance Initiative).

Table 32. Costs Of Incineration In Different Countries (€ / tonne MSW)

	Coopers and Lybrand et al (1997)				This Study				
					Incineration with en. Rec			Levies	
	Without energy recovery	With heat recovery	With power recovery	With heat and power recovery	Low	Med	High	Without energy recovery	With energy recovery
Austria					95		160		
Belgium	50	47	49	44					
Flanders						84		13	6
Denmark	98	88	86	77		43		44	37
Finland					Co-combusted / Gasification				
France	87	79	81	72	69		129		
Germany	103	93	97	86	90	120	250		
Greece	46	33	46	31	None				
Ireland	48	39	47	36	None				
Italy	48	30	33	18	100		200		
Luxembourg	104	95	97	88	97		121		
Netherlands	130	127	120	116	90		109		
Portugal	46	30	46	28	None				
Spain	46	35	46	33	18	30	51		
Sweden					37		87		
UK					41	51	66		
Cyprus					None				
Czech Republic					25		40		
Estonia					None				
Hungary									
Poland						60			
Slovenia					None				

5.4 Results for Composting

Costs of the composting process are likely to exhibit considerable variation. The key variable, in terms of process, is likely to be whether the process is carried out in-vessel, or in open air windrows. Pollution control equipment, front end separation technologies and techniques to reduce odours also affect the costs of the technology. Apart from the process costs, the costs of marketing are also an important consideration (though they are not considered in this section).

Open air windrow composting can cost less than €20 per tonne. This makes composting potentially one of the cheapest treatment technologies for municipal waste (see the Tables above for comparisons with landfill and incineration). However, in several situations, open-air windrowing of mixed biowastes might not be considered appropriate. The modelling of best practice technologies in this work (see Figure 13 and Table 33) suggests that a figure of €45 per tonne (excluding revenue) may be an appropriate figure for a plant of 20,000 tonnes capacity. Note that this best practice approach assumes quality separation of materials (i.e., greater than 95% purity) so that a primary screening step is not considered necessary.

A recent Austrian study suggested the figures in Table 34 below for composting. Similar figures apply to Germany, although an RVF report suggests much higher operating costs of €80-160 per tonne (Wannholt 1998). The same study gave a figure for a Danish plant of €106 per tonne, which again seems high. Figures for Greece (mixed waste) are at the lower end of this range, whilst lower tech approaches in Italy may be as low as €20 per tonne. Figures for high tech composting in Luxembourg appear to be relatively high at more than €70 per tonne.

Figure 13: Schematic Representation Of Best Practice, High-Tech Composting

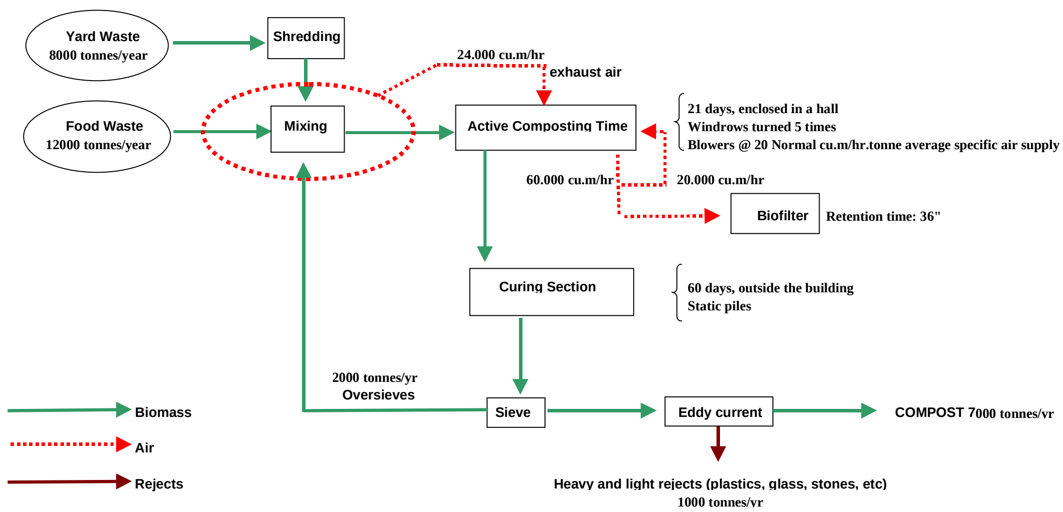


Table 33. Costs of 20,000tpa Composting Plant

Hi-Tech composting facility, 20.000 tpy

Brief Decription: **ACT**
phase in windrows on aerated floor (retention time: 21 days); exhaust air from the hall sucked and treated through a **Biofilter** (retention time: 36"). **Maturation** in static piles, outside the building (retention time: 60 days)

Capital Costs	Unit	Quantity	Unit cost, €	Cost	Depreciation time	Depreciation at 6%	Euro/tonne		Maintainai
Civil Works									
Paving, concrete	sq.m.	17000	35	595,000	20	- 51,875			1
Process Buidings	sq.m.	3950	150	592,500	20	- 51,657			1
Pool(s)	cu.m.	200	100	20,000	20	- 1,744			1
Biofilter	cu.m.	588	200	117,600	5	- 27,918			2
Weighing Bridge	#	1	30,000	30,000	10	- 4,076			2
Offices	cu.m.	300	300	90,000	20	- 7,847			2
Utilities	as a whole	1	300,000	300,000	10	- 40,760			5
Wall	m	600	100	60,000	20	- 5,231			1
			Total	**1,805,100**		**- 191,107**			
Equipment									
Shredder	#	1	150,000	150,000	8	- 24,155			5
Screw mixer	#	1	100,000	100,000	8	- 16,104			5
Turning Machine	#	1	250,000	250,000	8	- 40,259			5
Sieve	#	1	100,000	100,000	8	- 16,104			5
Eddy current separator	#	1	100,000	100,000	8	- 16,104			5
Loader	#	2	80,000	160,000	6	- 32,538			5
Hopper	#	1	30,000	30,000	8	- 4,831			5
Blowers, Fans	as a whole	1	250,000	250,000	8	- 40,259			5
			Total	**1,140,000**		**- 190,353**			
			TOTAL	**2,945,100**		- **381,461**	**-19.07**	-42.01 %	
			specific cost	147	€/tonne.year				

Operating costs		Unit	Quantity	Unit cost, €	Cost	Euros	Euro/tonne	
Manpower						**225,000**	**11.25**	24.78 %
	Director	#	1	50000	50,000			
	Accounter	#	1	35000	35,000			
	Operators	#	5	28000	140,000			
Fuels		litres	80,506	0.7	56,354	**56,354**	**2.82**	6.21 %
Energy		kWh	944,813	0.075	70,861	**70,861**	**3.54**	7.80 %
Maintainance		**see box "S28"**			89,427	**89,427**	**4.47**	9.85 %
Analysis		as a whole	1	25000	25,000	**25,000**	**1.25**	2.75 %
Disposal of rejects		tonnes/yea	1,000	60	60,000	**60,000**	**3.00**	6.61 %
				TOTAL	**908,103 €**		**45.41**	

Euro/tonne
45.41

	Type			Quantity		Sale price, €	
Sale of compost	Fresh compost		40%	2800	tonnes/yr		
end product: 7000 tonnes/yr		field crops	30%	2100		5	10,500
		horticulture	10%	700		40	28,000
	Ripe compost		60%	4200	tonnes/yr		
		bulk	40%	2800		15	42,000
		retail	20%	1400		40	56,000
							136,500 €

6.83

Table 34. Costs Of Composting, By Different Process, In Austria

Process	Feedstock	Unit Cost, €/t (and capacity)
Composting Plant	Garden- and Yardwaste, Source separated biowaste from households and commercial enterprises	670 €48 (20.000 t/a)
Biowaste-Sewage Sludge Co-Fermentation plus Composting Plant	Wet biowaste from households, Sewage Sludge, Zootechnical waste	325 €23 (20.000 t/a)
Home Composting	Garden- and Yardwaste, Source separated biowaste from households	-
Composting by Farmers	Liquid and solid animal manure, Garden- and Yardwaste, Source separated biowaste	450 – 780 €32 - €56

Source: Raninger et al (1998)

VROM quotes figures for the Netherlands of €54 per tonne, whilst the RVF study quoted above gives figures for operating costs at two Dutch plants of €40 and €46 per tonne. DHV (1997) quote figures for composting of €35-€60 per tonne, whilst another source from Germany suggests a range €80-90 (cited in Tobin Environmental Services 1999). A review of technologies carried out by NOVEM (1992) gives ranges for the treatment costs per tonne depending upon capacity. The figures range from 168 NLG (approx €80) per tonne at low capacities (10,000 tonnes) to between 61-123 NLG (approx €30-€60) per tonne (depending upon technology) at the higher capacity end (around 50,000 tonnes). The same study gives figures between 77NLG-86NLG (approx €38-€43) for 25,000 tonne plant, which is consistent with the estimate made in this study, though the two estimates for 20,000 tonne plant are much higher (142-154 NLG, or €71-€77 per tonne) probably reflecting the technologies used.

RVF quotes a wide range of costs for Sweden (€34 - €90), but with costs expected to fall (whilst costs of other treatments are on the increase). In Slovenia, the operational cost for the two composting plants of Koper and Maribor are € 28-31 /tonne of compost produced and € 22 /tonne of compost produced, respectively. The operational cost for the composting plant in Vrhnika is 18 Euro/tonne of waste treated.[24] A recent UK review suggested costs of windrow composting were between €16-32 (Composting Association 1997). Vertical in-vessel units are now available and being used in the UK (as well as the United States and New Zealand), the costs for which are of the order €32 per tonne (at 20,000 tonnes capacity).[25]

Much attention is paid to cross country variation in costs. For composting, it would seem that the cost variation within countries, through choice of technology and scale

[24] Information from Member State Consultations.

[25] Communication with Orrtec.

of plant, is likely to be at least as large as the variation in costs across countries for an established technology. Indeed, cost data from 'best practice' composting in Southern Member States was circulated to process engineers and consultants in Northern Member States and in some cases, the latter quote lower figures for comparable treatments. Much clearly depends on the detail of the process (capital equipment, separation, aeration, maturation period etc.) and the assumptions regarding the composition of input wastes.

In each of the above cases, it is assumed that the process is being applied to source segregated municipal waste. It is well known that attempts to compost mixed municipal waste (i.e. waste that is not collected at source) leads to much higher levels of contamination with various materials, including heavy metals. These contaminants make the material less suitable for useful applications. Well-defined standards assist in establishing the fitness-for-purpose of different grades of compost.

In this study, the figures used are representative of good practice composting incorporating biofilters to reduce emissions (so the costs of green waste composting in windrows may be over-estimated). The figures incorporate an estimated sales value of €6-7 per tonne of waste based on typical marketing mixes and sales prices. The net cost (after sales) is of the order €40 per tonne. This assumes a €60 per tonne cost for disposal of rejects (accounting for 6.6% of total costs). Manpower, fuels and energy will vary in cost across Member States. Fuels and energy comprise 14% of the annualised per tonne costs whilst manpower accounts for 25% of costs. If, for the purposes of sensitivity analysis, one doubles the cost of each of these variable items, the net per tonne cost increases to €60. Equally, there would be reason to believe that the unit labour costs used in the study are well above what would apply in some of the countries under consideration, notably the Accession States.

These costs can be compared with those given by the European Topic Centre for Waste (2001). The figures quoted in that study are somewhat higher. The operating costs of a 20,000 tonne compost plant with forced aeration are equivalent to €80 per tonne. Even the higher estimates discussed above suggest this is very high indeed. If one adds an estimate of the annualised capital costs, the per tonne costs net of revenue appear to be of the order €130 per tonne, which seems far too high.

Given that plant capacities are likely to vary across different situations, 'high' and 'low' costs per unit of waste composted are estimated. These figures are based upon the 'best practice model' described above, though equally, the variation in technologies available could lead to much higher, or (depending upon the situation) much lower costs. The range used is €35-60 for best practice plant.

5.5 Results for Anaerobic Digestion

For anaerobic digestion, different approaches to treatment will have different costs. Similarly, since some processes incorporate co-digestion of wastes, the costs may be affected by the materials which are treated alongside the source-separated MSW. The way in which these affect the costs of the system are likely to depend upon the way in which they affect other cost factors such as the need for pre-treatment of the wastes, the need for waste waster treatment, the quality of the digestate, the amount of biogas produced and its energy content, and revenues generated from sales of energy (and whether both electricity and thermal energy are sold). These factors will themselves be influenced by different regulations in different countries. As with incineration, subsidy regimes for renewable energy may have an important role to play.

Few countries have operating anaerobic digestion (AD) plants in place for the treatment of municipal wastes. For Austria, figures are given in Table 35. Additional figures from a study by RVF are given in Table 36.

Table 35. Costs Of Anaerobic Digestion Of Different Materials In Austria

Process	Feedstock	Unit Cost, €/t (and capacity)
Biowaste Fermentation plus Composting Plant	Source separated biowaste from households and commercial enterprises, Foodwaste	(€80) (20.000 t/a)
Biowaste-Sewage Sludge Co-Fermentation plus Composting Plant	Wet biowaste from households, Sewage Sludge, Zootechnical waste	(€23) (20.000 t/a)

Source: Raninger et al (1998)

Table 36. Operating Costs Of Anaerobic Digestion Plants In Different Countries

Country	NL	GER	B	FIN	F	S
Location	Tilburg	Baden-Baden	Brecht	Vaasa	Amiens	Sobacken / Boras
Per tonne costs (€)	139.82	159.01	65.94	48.55	55.75	56.51

Source: Wannholt (1998)

A recent UK study suggests costs could lie between €80-€96 net of electricity sales estimated at approximately €5-€8/tonne. This report suggests that though the capital costs fall with scale, the efficiency of energy production falls with increasing throughput (Waterman BBT 1999). Some European schemes appear to generate significant revenue from the sale of digestates since these may have significant nutrient values (IWM Anaerobic Digestion Working Group 1998). A review by NOVEM (1992) suggested net treatment costs, including electricity supply, as high as €130 for small systems (10,000 tonnes) but falling to around €50 for systems of capacity 50,000 tonnes or so. However, the different systems examined exhibited some variation in costs.

The review above has identified a number of different variables which alter anaerobic digestion's costs, environmental impacts and end-product quality. This study must therefore identify best practice from the available data to model the impacts of a future AD plant. It is important to note however that because of the flexibility inherent in the technology, it can effectively model itself around local priorities and pressures which may vary significantly, for example, requirements for energy recovery, complete removal of pathogens, specific feedstock types, etc. Furthermore, what is appropriate for urban environments will not be appropriate for rural areas and so best practice ought to distinguish between the two locations. The situation will also differ between Northern and Southern Europe. The following best practice model will address the process characteristics, separation techniques and plant capacity according to the urban and rural scenarios. It will also examine probable differences between the situations in Northern and Southern Europe.

Process characteristics

In general, the analysis above highlighted the advantages attributed to thermophilic digestion in terms of pathogen destruction, biogas production rate, retention time, etc. Therefore in spite of it being comparatively underused in comparison with mesophilic digestion (62% of AD plants operate at mesophilic temperatures across Europe), this study will consider the thermophilic process as best practice as it combines hygienic aspects with high rates of digestion.

Urban digestion

There are a number of important factors associated with urban areas relevant to digestion. A high biogas production would be favourable as it would lead to a lower digestate output (digestate is not as important an end-product in more populated areas). This high biogas production subsequently leads to more energy recovery in terms of electricity and also heat because a higher population density means more can be utilised. Therefore, this supports the argument for thermophilic digestion. It should be noted that as mentioned above, in general larger centralised plants require larger proportions of their generated electricity for plant operation. In terms of capacity, the argument here is for larger centralised plants, compared with rural areas, which maximise cost efficiency (plant and collection) and serve a higher percentage of the population, thus diverting a significant portion of biowaste from landfill.

Rural digestion

Classically anaerobic digestion is more widespread in rural areas where the needs are quite different. AD can be a valuable process for farmers in particular, in terms of additional income (end-product sales), reduced odour dispersion in comparison with the spreading of manure, and digested manure is more homogenous and easier to apply. There will be a much higher demand for digestate for agricultural or horticultural purposes compared with urban areas and whilst that might contradict the arguments articulated above, end-product quality is a key factor. The higher-temperature technology ensures a higher destruction of potentially toxic materials such as pathogens which make the digestate more useful, easier to handle and more hygienic in general. Whilst energy recovery is an important aspect, it is doubtful that all the energy (heat and electricity) could be utilised on site. Smaller capacities are more logical here because of the lower population densities and because of the possibilities of farm-based AD plants. This also raises the option of co-digestion (also potentially important for urban plants) with other wastes such as manure, an important factor but one which cannot be modelled in this study. However, AD's flexibility in co-digesting waste from agricultural origin with municipal biowaste can make it preferable compared with composting.

Important developments on this side are currently being reported for example in Northern Italian Alpine Regions, where local regulations promote farm-scale AD facilities to treat manure and source separated food waste.

Costs are detailed for centralised AD using source separated MSW in more urbanised environments (see Table 37) and for on-farm digestion in rural areas (see Table 38).

Table 37. Costs For Best Practice Scenario Centralised Plant For The Treatment Of 15,000 Tons Biodegradable Waste Per Year

		Centralised AD
Processing capacity		15.000 tonne biowaste per annum
Capital cost	interest rate 7.5 %, operating time 10 years	70 Euro per tonne biowaste processing capacity
Operating cost		48 Euro per tonne biowaste
Value of outputs		
Surplus Electricity	0.075 Euro/kWh	3.8 Euro per tonne biowaste
Heat	0.02 Euro/kWh (assuming that 40 % could be used)	2.4 Euro per tonne biowaste
Compost	10 Euro per tonne produced	3 Euro per tonne biowaste
Net cost		**108.8 Euro/ton**

Source: ZREU (2000)

Table 38. Cost Data For On-Farm Anaerobic Digestion

		Farm AD
Plant capacity		2,500 tonne biowaste per annum
Capital cost	interest rate 7.5 %, operating time 10 years	30 Euro per tonne biowaste processing capacity
Operating cost		60 Euro per tonne biowaste
Value of outputs		
Surplus Electricity	0.075 Euro/kWh	9 Euro per tonne biowaste
Heat	0.02 Euro/kWh (assuming that 25 % could be used as added value)	1.5 Euro per tonne biowaste
Net cost		**79.5 Euro/tonne**

Source: ZREU (2000)

For centralised digestion, source-separated MSW is used and costs are higher at €108.8 per tonne. The on farm figures are based on a German plant the costs for which are €79.5 for the treatment. Note that both include a post-digestion composting phase which is not typical in Denmark and Sweden (where digestate tends to be applied direct to land). Swedish figures suggest that a 15-20,000 tonne digester would have a gate fee of €60-70 excluding a post-digestion composting phase. This suggests the costs of the centralised digestion costs estimated above are in the right area for one which includes a composting stage to stabilise the digestate.

The best practice system used in centralised systems is assumed to operate with a post-digestion composting phase. In some countries where the process is controlled by farmers (such as Denmark) the digested material is applied without going through this type of stabilisation phase. This may reduce costs.[26] The cost data used in this study are €80-110. Again, this can be compared with the costs quoted in the recent European Topic Centre for Waste (2001) report. There, operating costs are very low indeed, whilst capital costs are lower than for compost plants of the same capacity. This has the effect of making anaerobic digestion appear a cheaper technology than composting, even without forced aeration. Indeed, the operating costs for composting without forced aeration are more than seven times those for quoted the equivalent size anaerobic digestion plant. The ETCW figures appear low unless it is being assumed that extremely high revenues are generated from energy sales, or if it is assumed that the aerobic phase is always omitted (with delivery of digestate direct to farmers, so avoiding also the costs of waste water treatment).

[26] Simon Lundberg of the Swedish EPA points out that in Sweden, the costs are between 60-70€ per tonne where wet waste is applied to land.

6.0 RESULTS OF THE ECONOMIC ANALYSIS

6.1 Financial and Economic Costs and Benefits of Separate Collection

Bringing together the data concerning environmental and economic costs and benefits of different waste management options is a process full of pitfalls. It is difficult enough to find or generate reliable quantitative data on either aspect of any individual treatment option. As shown in Section 4, assessment of the external costs of each of the treatments is inevitably incomplete given gaps in relevant scientific and economic relationships. Section 5 illustrates that it is not only the external costs which are subject to some conjecture. Within any given country, private costs for treatments also exhibit variation in accordance with local factors, scale, choice of technology, etc. In this context, it is quite a hazardous enterprise to seek to amalgamate the two sets of data.

As a minimum, one ought to have some confidence that the environmental costs and the financial costs one is using at least relate to the same process. This is less straightforward than it might at first appear. First of all, not all incinerators in Europe currently meet all the standards laid down in the most recent Incineration Directive, yet in the external cost modelling, it was effectively assumed that all of them do. Where incinerators do not meet these standards, costs will be incurred in bringing them up to these standards. Equally, where incinerators are routinely well within such limits, the environmental costs may be overstated relative to the actual situation.

The equivalent problem in respect of landfill is especially awkward to deal with. In the external cost analysis, it is assumed that landfills would be ‘well-behaved’, on the basis that the time taken to introduce a policy on separate collection would probably be such that the ramifications of the Landfill Directive would be being felt in the countries concerned. However, the financial cost data relate, in some cases, to landfills which are a long way from meeting the requirements of the Landfill Directive. As such, for many countries, the financial costs reported are almost certainly well below those which would prevail if landfills of the standards assessed in this document existed in all countries. Furthermore, as diversion of material away from landfill proceeds under the Landfill Directive, the fill rates for landfills will decline, putting upward pressure on gate fees (other things being equal). There is, therefore, a dynamic in the process which is extraordinarily difficult to capture and which confounds any attempt to make sensible forward projections.

Having said that, to the extent that the Landfill Directive is concerned with diversion of materials away from landfill, it could be argued that the key issues are how the diversion of material from landfill evolves over time. A policy requiring the separate collection of biowastes will tend to shift the balance of diversion in favour of composting and anaerobic digestion and away from alternatives such as incineration (and pyrolysis, gasification, mechanical biological treatment etc., which are not considered in this study). Even here, the future for treatments such as MBT and the more modular scale thermal treatments is difficult to estimate. Certainly, it would appear that mass-burn incineration plants would become less favoured, whilst the remaining residual fractions could be treated by combinations of MBT and anaerobic digestion, pyrolysis and gasification.

For the financial cost data, the figures from Chapter 5 were used. This suggests that the separate collection itself will cost an additional €0-15 per tonne (low and high additional costs under best practice).

The figures used for compost and anaerobic digestion are given in Table 39 below. These are representative of variation in choice of technology, variation in costs of disposal of rejects (which vary locally as much as they do nationally), the income derived from sales of compost etc. For anaerobic digestion, assuming that the digestate is de-watered and stabilised through an aerobic composting phase prior to being utilised as compost, figures of €80-110 were used.

Table 39. Financial Costs Used for Composting and Anaerobic Digestion

Process	Low (€/t)	High (€/t)
Separate Collection (increment above standard residual collection, expressed per tonne biowaste collected)	0	15
Composting	35	60
Anaerobic Digestion	80	110
Separate Collection Plus Composting	35	75
Separate Collection Plus Anaerobic Digestion	80	125

6.1.1 A Note on Methodology

The approach taken in deriving the results which follow is to look at combinations of scenarios where the results of the externality analysis, using both high and low unit damage cost, are combined with the low and high cost figures for the different treatments. This gives broad ranges for the results. This is an appropriate approach rather than assuming that one can know what the 'best' estimate of the external or private costs actually is in the given situation. The fact remains that without better knowledge of the specific situations, positing so-called best estimates, or mid-point estimates, can be misleading since these will frequently be taken as 'the answer' whilst the more illuminating analysis concerning the ranges which may exist becomes lost.

The future private costs of the treatment options, and the external costs associated with each of those options, are both difficult to predict. For the most part the external costs of each option should fall as private costs increase.

6.2 Switching from Landfill to Composting

Table 40 below shows the external benefits, the private costs and the total economic cost of the switch from landfill to incineration. The figures in Table 40 show externalities for the landfills operating at currently estimated gas collection efficiencies and private costs estimated at the current level. Landfill taxes are included in the final row although, arguably, doing so implies a degree of double counting of costs where such taxes are intended to reflect the external costs of landfilling.

Table 40. Net External Costs And Private Costs Of Change From Landfill To Composting (€ / tonne of waste switched, current landfill practice)

		AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																							
Raw Score (direct emissions only) = A		10.2	13.4	6.8	10.2	3.8	6.4	4.3	7.3	5.1	8.2	3.9	6.7	9.9	13.8	9.4	13.2	5.7	8.9	6.8	9.8	4.9	7.8
plus Avoided External Costs Energy = B		10.1	13.2	6.5	9.5	3.1	5.0	4.0	6.6	4.3	7.1	3.3	5.6	9.9	13.7	9.2	12.9	5.2	8.0	6.7	9.2	4.6	6.9
plus Disamenity = C		12.1	17.2	13.6	23.7	6.1	10.9	4.3	7.3	6.5	11.5	8.6	16.3	11.3	16.5	10.1	14.6	8.9	15.3	10.0	15.9	13.1	23.8
plus Avoided Private Costs Fertiliser = D		12.9	19.2	14.4	25.6	6.8	12.9	5.1	9.2	7.3	13.4	9.4	18.2	12.0	18.4	10.8	16.5	9.6	17.2	10.8	17.8	13.9	25.8
plus Avoided Private Costs of Pesticides = E		13.3	19.6	15.8	27.1	7.0	13.0	5.2	9.3	7.9	14.0	9.7	18.5	12.5	18.9	12.6	18.3	10.6	18.2	11.3	18.2	15.3	27.2
Private Costs																							
Landfill = F	Low	55.0	55.0	52.0	52.0	44.0	44.0	28.0	28.0	31.0	31.0	35.0	35.0	5.0	5.0	34.0	34.0	50.0	50.0	123.0	123.0	75.0	75.0
	High	110.0	110.0	56.0	56.0	44.0	44.0	56.0	56.0	85.0	85.0	220.0	220.0	21.0	21.0	78.0	78.0	70.0	70.0	147.0	147.0	75.0	75.0
Compost plus Separate Collection = G	Low	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0
	High	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0
Net Private Cost = (G – F) = H	Low	-20.0	-20.0	-17.0	-17.0	-9.0	-9.0	7.0	7.0	4.0	4.0	0.0	0.0	30.0	30.0	1.0	1.0	-15.0	-15.0	-88.0	-88.0	-40.0	-40.0
	High	-35.0	-35.0	19.0	19.0	31.0	31.0	19.0	19.0	-10.0	-10.0	-145.0	-145.0	54.0	54.0	-3.0	-3.0	5.0	5.0	-72.0	-72.0	0.0	0.0
Totals																							
Costs Net of External Benefits, Raw Score =H - A	Low	-30.2	-33.4	-23.8	-27.2	-12.8	-15.4	2.7	-0.3	-1.1	-4.2	-3.9	-6.7	20.1	16.2	-8.4	-12.2	-20.7	-23.9	-94.8	-97.8	-44.9	-47.8
	High	-45.2	-48.4	12.2	8.8	27.2	24.6	14.7	11.7	-15.1	-18.2	-148.9	-151.7	44.1	40.2	-12.4	-16.2	-0.7	-3.9	-78.8	-81.8	-4.9	-7.8
Costs Net of External Benefits, Including Avoided External Costs of Energy = H - B	Low	-30.1	-33.2	-23.5	-26.5	-12.1	-14.0	3.0	0.4	-0.3	-3.1	-3.3	-5.6	20.1	16.3	-8.2	-11.9	-20.2	-23.0	-94.7	-97.2	-44.6	-46.9
	High	-45.1	-48.2	12.5	9.5	27.9	26.0	15.0	12.4	-14.3	-17.1	-148.3	-150.6	44.1	40.3	-12.2	-15.9	-0.2	-3.0	-78.7	-81.2	-4.6	-6.9
Costs Net of External Benefits, Including Avoided External Costs of Energy and Landfill Tax = H-B + landfill tax	Low	-73.1	-76.2	-75.5	-78.5	-62.1	-64.0	-12.0	-14.6	-9.3	-12.1	-3.3	-5.6	20.1	16.3	-30.2	-33.9	-30.2	-33.0	-94.7	-97.2	-104.6	-106.9
	High	-88.1	-91.2	-42.5	-42.5	-22.1	-24.0	0.0	-2.6	-23.3	-26.1	-148.3	-150.6	44.1	40.3	-34.2	-37.9	-50.2	-53.0	-78.7	-81.2	-64.6	-66.9

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
External Benefits		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																					
Raw Score (direct emissions only) = A		8.06	9.16	7.45	8.42	3.81	4.73	3.41	4.04	9.72	10.91	9.72	10.91	9.72	10.90	8.90	10.07	9.74	10.99	8.91	10.12
plus Avoided External Costs Energy = B		7.96	8.90	7.28	8.08	3.50	4.01	2.88	2.97	9.66	10.79	9.66	10.79	9.65	10.78	8.83	9.95	9.68	10.87	8.85	9.99
plus Disamenity = C		9.81	12.61	8.55	10.62	4.06	5.14	8.09	13.39	10.91	13.30	11.68	14.84	10.17	11.82	10.50	13.27	11.61	14.74	10.35	13.00
plus Avoided Private Costs Fertiliser = D		10.59	14.54	9.32	12.55	4.83	7.05	8.85	15.30	11.67	15.21	12.45	16.75	10.94	13.73	11.27	15.20	12.38	16.65	11.12	14.93
plus Avoided Private Costs of Pesticides = E		11.23	15.18	9.57	12.79	4.95	7.18	9.54	15.99	13.15	16.69	13.93	18.23	11.13	13.92	11.74	15.67	13.82	18.10	11.77	15.58
Private Costs																					
Landfill = F	Low	6	6	6	6	22	22	10	10	5	5	15	15	15	15	15	15	15	15	5	5
	High	15	15	30	30	82	82	34	34	21	21	25	25	25	25	25	25	25	25	5	5
Compost plus Separate Collection = G	Low	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35	35
	High	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75	75
Net Private Cost = (G – F) = H	Low	29	29	29	29	13	13	25	25	30	30	20	20	20	20	20	20	20	20	30	30
	High	60	60	45	45	-7	-7	41	41	54	54	50	50	50	50	50	50	50	50	70	70
Totals																					
Costs Net of External Benefits, Raw Score =H - A	Low	20.94	19.84	21.55	20.58	9.19	8.27	21.59	20.96	20.28	19.09	10.28	9.09	10.28	9.10	11.10	9.93	10.26	9.01	21.09	19.88
	High	51.94	50.84	37.55	36.58	- 10.81	- 11.73	37.59	36.96	44.28	43.09	40.28	39.09	40.28	39.10	41.10	39.93	40.26	39.01	61.09	59.88
Costs Net of External Benefits, Including Avoided External Costs of Energy = H - B	Low	21.04	20.10	21.72	20.92	9.50	8.99	22.12	22.03	20.34	19.21	10.34	9.21	10.35	9.22	11.17	10.05	10.32	9.13	21.15	20.01
	High	52.04	51.10	37.72	36.92	- 10.50	- 11.01	38.12	38.03	44.34	43.21	40.34	39.21	40.35	39.22	41.17	40.05	40.32	39.13	61.15	60.01
Costs Net of External Benefits, Including Avoided External Costs of Energy and Landfill Tax = H-B + landfill tax	Low	21.0	20.1	21.7	20.9	-20.5	-21.0	3.1	3.0	20.3	19.2	9.3	8.2	10.3	9.2	11.2	10.1	10.3	9.1	21.2	20.0
	High	52.0	51.1	37.7	36.9	-40.5	-41.0	19.1	19.0	44.3	43.2	39.3	38.2	40.3	39.2	41.2	40.1	40.3	39.1	61.2	60.0

6.2.1 External Benefits

These vary across countries depending upon the estimates of gas collection efficiency. For countries with landfills which perform well in this respect (e.g. Germany, United Kingdom), external benefits can be as low as €3-4 per tonne. Where landfills perform less well (e.g. Accession States, Greece), benefits are of the order €10 per tonne. It almost certainly the case that this analysis does not pick up all the benefits of the switch owing to the incomplete nature of the analysis.

In future, the performance of landfills in the countries where performance is less good will improve. This will increase costs and reduce the benefits of the switch. Hence, in Table 41, the results are shown for the case in which all countries have landfills which perform well. The landfills are characterised by the following parameters:

Oxidation rate	10%
Landfill gas capture	10%
Used for energy recovery	0%

The private costs of landfilling are estimated to rise to €55 in all countries where gate fees are currently estimated below this.

In this case, the external benefits of the switch are much lower. This is due to the fact that in the relatively high performance landfill being posited, a lot of methane is captured and converted to carbon dioxide. The process of methanogenesis is effectively occurring over a more extended period of time (in the modelling) than in compost, where much of the carbon is liberated swiftly as carbon dioxide in the composting process. This means that carbon dioxide and methane emissions (which are reduced owing to oxidation and high capture rates for flaring / energy recovery) from landfill are reduced by the effect of discounting. The same is true to some extent of composting, but only for the carbon dioxide emissions from the organic matter in the composted material.

The analysis, such as it stands, is also significantly affected by how one treats the production of energy from landfill gas, and on the external costs associated with energy use at the composting site. The greater both of these become, the more the benefits associated with switching to composting and away from landfill (as estimated here) fall. Indeed, the net benefits can switch negative, as indicated in the results in the Table.

6.2.2 Private Costs

Unsuprisingly, in those countries where most headway has been made with composting, the net private costs of the switch are negative. Arguably, this explains the progress made. It should be made clear, however, that whether or not a landfill tax is in place is not the key factor in ensuring the switch can occur at negative cost. This itself is an important observation since it suggests a need to understand the variation in private costs better than the present state of knowledge allows.

Table 41. Net External Costs And Private Costs Of Change From Landfill To Composting (€ / tonne of waste switched, good practice landfill, increased costs)

		AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																							
Raw Score (direct emissions only) = A		1.7	4.0	2.0	4.9	1.5	3.9	1.9	4.7	0.3	2.9	1.5	4.1	1.3	4.2	1.8	4.9	0.9	3.6	2.1	4.6	2.5	5.2
plus Avoided External Costs Energy = B		0.9	2.3	1.3	3.5	0.5	1.9	1.5	3.7	-1.3	0.7	0.7	2.6	0.8	3.4	0.4	3.0	0.0	1.9	1.7	3.3	2.2	4.0
plus Disamenity = C		3.0	6.4	8.4	17.8	3.5	7.9	1.8	4.4	0.9	5.1	6.1	13.3	2.2	6.2	1.3	4.7	3.6	9.2	5.1	10.1	10.6	20.9
plus Avoided Private Costs Fertiliser = D		3.7	8.3	9.2	19.7	4.2	9.8	2.6	6.4	1.7	7.0	6.8	15.2	3.0	8.1	2.1	6.6	4.4	11.1	5.9	12.0	11.4	22.8
plus Avoided Private Costs of Pesticides = E		4.2	8.7	10.7	21.1	4.4	10.0	2.7	6.5	2.3	7.6	7.1	15.5	3.4	8.6	3.8	8.4	5.4	12.1	6.3	12.4	12.8	24.3
Private Costs																							
Landfill = F	Low	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	123.0	123.0	75.0	75.0
	High	110.0	110.0	56.0	56.0	55.0	55.0	56.0	56.0	85.0	85.0	220.0	220.0	55.0	55.0	78.0	78.0	70.0	70.0	147.0	147.0	75.0	75.0
Compost plus Separate Collection = G	Low	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0
	High	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0
Net Private Cost = (G – F) = H	Low	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-20.0	-88.0	-88.0	-40.0	-40.0
	High	-35.0	-35.0	19.0	19.0	20.0	20.0	19.0	19.0	-10.0	-10.0	-145.0	-145.0	20.0	20.0	-3.0	-3.0	5.0	5.0	-72.0	-72.0	0.0	0.0
Totals																							
Costs Net of External Benefits, Raw Score =H - A	Low	-21.7	-24.0	-22.0	-24.9	-21.5	-23.9	-21.9	-24.7	-20.3	-22.9	-21.5	-24.1	-21.3	-24.2	-21.8	-24.9	-20.9	-23.6	-90.1	-92.6	-42.5	-45.2
	High	-36.7	-39.0	17.0	14.1	18.5	16.1	17.1	14.3	-10.3	-12.9	-146.5	-149.1	18.7	15.8	-4.8	-7.9	4.1	1.4	-74.1	-76.6	-2.5	-5.2
Costs Net of External Benefits, Including Avoided External Costs of Energy = H - B	Low	-20.9	-22.3	-21.3	-23.5	-20.5	-21.9	-21.5	-23.7	-18.7	-20.7	-20.7	-22.6	-20.8	-23.4	-20.4	-23.0	-20.0	-21.9	-89.7	-91.3	-42.2	-44.0
	High	-35.9	-37.3	17.7	15.5	19.5	18.1	17.5	15.3	-8.7	-10.7	-145.7	-147.6	19.2	16.6	-3.4	-6.0	5.0	3.1	-73.7	-75.3	-2.2	-4.0
Costs Net of External Benefits, Including Avoided External Costs of Energy and Landfill Tax = H-B + landfill tax	Low	-63.9	-65.3	-73.3	-75.5	-70.5	-71.9	-36.5	-38.7	-27.7	-29.7	-20.7	-22.6	-20.8	-23.4	-42.4	-45.0	-30.0	-31.9	-89.7	-91.3	-102.2	-104.0
	High	-78.9	-80.3	-37.3	-36.5	-30.5	-31.9	2.5	0.3	-17.7	-19.7	-145.7	-147.6	19.2	16.6	-25.4	-28.0	-45.0	-46.9	-73.7	-75.3	-62.2	-64.0

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																					
Raw Score (direct emissions only) = A		0.4	0.8	0.3	0.5	1.4	2.1	1.0	1.4	1.0	1.4	1.0	1.4	1.0	1.4	0.2	0.5	1.0	1.4	0.2	0.6
plus Avoided External Costs Energy = B		-0.1	-0.6	-0.5	-0.9	1.0	1.2	0.3	0.0	0.3	-0.1	0.3	-0.1	0.3	-0.1	-0.5	-0.9	0.3	0.0	-0.5	-0.9
plus Disamenity = C		1.7	3.1	0.8	1.6	1.6	2.3	5.5	10.4	1.5	2.4	2.3	3.9	0.8	0.9	1.1	2.4	2.2	3.8	1.0	2.1
plus Avoided Private Costs Fertiliser = D		2.5	5.0	1.6	3.5	2.4	4.2	6.3	12.3	2.3	4.3	3.1	5.9	1.6	2.8	1.9	4.3	3.0	5.8	1.7	4.0
plus Avoided Private Costs of Pesticides = E		3.1	5.7	1.8	3.8	2.5	4.4	7.0	13.0	3.8	5.8	4.5	7.3	1.7	3.0	2.4	4.8	4.4	7.2	2.4	4.7
Private Costs																					
Landfill = F	Low	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0
	High	55.0	55.0	55.0	55.0	82.0	82.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0
Compost plus Separate Collection = G	Low	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0
	High	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0
Net Private Cost = (G – F) = H	Low	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	-20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0	- 20.0
	High	20.0	20.0	20.0	20.0	-7.0	-7.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0
Totals																					
Costs Net of External Benefits, Raw Score =H - A	Low	- 20.4	- 20.8	- 20.3	- 20.5	- 21.4	- 22.1	- 21.0	- 21.4	- 21.0	- 21.4	-21.0	- 21.4	- 21.0	- 21.4	- 20.2	- 20.5	- 21.0	- 21.4	- 20.2	- 20.6
	High	19.6	19.2	19.7	19.5	-8.4	-9.1	19.0	18.6	19.0	18.6	19.0	18.6	19.0	18.6	19.8	19.5	19.0	18.6	19.8	19.4
Costs Net of External Benefits, Including Avoided External Costs of Energy = H - B	Low	- 19.9	- 19.4	- 19.5	- 19.1	- 21.0	- 21.2	- 20.3	- 20.0	- 20.3	- 19.9	-20.3	- 19.9	- 20.3	- 19.9	- 19.5	- 19.1	- 20.3	- 20.0	- 19.5	- 19.1
	High	20.1	20.6	20.5	20.9	-8.0	-8.2	19.7	20.0	19.7	20.1	19.7	20.1	19.7	20.1	20.5	20.9	19.7	20.0	20.5	20.9
Costs Net of External Benefits, Including Avoided External Costs of Energy and Landfill Tax = H-B + landfill tax	Low	- 19.9	- 19.4	- 19.5	- 19.1	- 51.0	- 51.2	- 39.3	- 39.0	- 20.3	- 19.9	-21.3	- 20.9	- 20.3	- 19.9	- 19.5	- 19.1	- 20.3	- 20.0	- 19.5	- 19.1
	High	20.1	20.6	20.5	20.9	- 38.0	- 38.2	0.7	1.0	19.7	20.1	18.7	19.1	19.7	20.1	20.5	20.9	19.7	20.0	20.5	20.9

In many respects, it should not be so surprising that landfills might have higher costs than composting plants. The emissions from composting are, in the main, more benign than those from landfills, which are also likely to be less easily controlled. The potential for accidents at landfill sites, as well as long-term, irreversible damage to groundwaters, may justify measures to control landfill sites more closely, though the external benefits derived from this are extremely difficult to quantify. This in turn relates to the nature of the materials being treated. Source separated biowastes, where the process is well-managed, can be dealt with in the absence of major risks whilst landfills, because of the heterogeneity of input wastes, have to be designed to cope with a range of materials. The composting process takes place over a relatively short period of time, whereas the costs of landfilling have much to do with aftercare once the material has been made more or less inaccessible (especially following implementation of the Landfill Directive).

It is difficult to look at the above Table without passing comment on the enormous variation in the net private costs across countries. This is a reflection of the variation in landfill gate fees across the countries being assessed. This variation is deserving of a more detailed study in its own right to seek to understand the source of the variation. It is to be expected that these would converge over time, but there is also good reason to believe that complete convergence is a long way off. Several countries and / or regions now have, or are soon to implement, bans on specific wastes, or, as in Austria and Germany, bans on landfilling of wastes without first pre-treating them.

6.2.3 Costs Net of External Benefits

The costs net of external benefits are shown to shed some light upon whether the external benefits generated are worth paying for. A negative figure suggests that the benefits exceed the costs of the change. Where the costs are positive, this suggests that the ratio of quantifiable benefits to costs is less than one. In theory, this implies the change is not something worthy of current investment. In this case, however, the incomplete nature of the analysis makes such statements difficult to make with any certainty.

Table 40 shows that the net cost-benefit balance is not negative for al countries. Figures are positive for Greece, Portugal, Spain, the UK and the Accession States. Table 41 indicates that in the scenario in which the landfills perform well, but where the costs are higher, the countries for which the balance is negative are Finland, Greece, Portugal, Spain and the Accession States, but only in the high cost scenario. This shows that for this scenario, the benefits from the switch fall, but in the low cost scenario the costs of the switch are lower than the costs for separate collection and composting.

The key issues seem to be a) whether the landfill gate fees are greater or less than the incremental costs of source separation plus the gate fee for composting in future; and b) the degree to which the omissions on the externality assessment affect the analysis. The former question will be influenced by how much the costs for landfilling will increase as a consequence of implementation of the Landfill Directive (since the external costs and benefits are assessed with respect to a landfill performing at 'post-implementation' levels).

6.2.4 Comment on Landfill Taxes

Looking at the differential external costs of landfill and composting, one could look at the level of landfill taxes and argue that they are too high (given the absence of similar instruments applied to composting). This is less than clear.

First of all, the incompleteness of the externality assessments cautions against such statements. In the case of landfill, the externalities left unquantified are without exception 'negative externalities'. Secondly, the orthodox interpretation implied by such a comment suggests that there is no room for alternative rationales for the existence of landfill taxes. Different countries have different reasons for introducing such instruments. Arguably, landfill / waste taxes are very useful revenue raising instruments from the perspective of Treasuries. More likely, the design of such taxes is intended to a) encourage waste minimisation and b) encourage a shift in waste management up the waste management hierarchy. In this context, it is worth noting that a number of studies have suggested that the net external benefits of recycling a typical basket of recyclables vis a vis compostables would be of the order €100 and more. As a means to encourage recycling, therefore, and source separation more generally, landfill / waste taxes are not devoid of utility and are not excessive.

6.3 Switching from Landfill to Anaerobic Digestion

Tables 42 and 43 below shows the external benefits, the private costs and the total economic costs of the switch from landfill to anaerobic digestion. Again, this is shown for current practice landfilling (Table 42) and for the case in which landfills perform better and are higher in cost (Table 43).

6.3.1 External Benefits

Measured with reference to the raw scores, the external benefits associated with this switch are generally similar to those for the switch from landfill to composting. They also follow the same pattern as indicated above for composting. However, once one introduces avoided burdens associated with energy generated from AD, the external benefits move more strongly in favour of AD.

Again, as with the switch from landfill to compost, the external benefits are much smaller where landfills perform better. There is, however, no case where the benefits are negative under either scenario. This is due to the external benefits of energy production.

6.3.2 Private Costs

Despite the larger external benefits from AD, these come at a considerably increased price. Hence, the net private costs are much less favourable than in the switch from landfilling to composting. Indeed, in Table 43 (where landfill costs are high) the costs of the switch are positive in most cases.

Table 42. Net External Costs And Private Costs Of Change From Landfill To AD (€ / tonne of waste switched, current landfill practice)

		AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																							
Raw Score (direct emissions only) = A		10.5	14.2	7.3	10.7	4.2	7.1	4.2	7.0	6.2	9.6	4.3	7.3	9.8	13.5	10.1	13.8	6.1	9.6	6.8	9.9	4.8	7.9
plus Avoided External Costs Energy (from landfill = B)		10.5	14.0	6.9	9.9	3.5	5.7	3.8	6.3	5.4	8.5	3.7	6.2	9.8	13.5	9.9	13.5	5.6	8.8	6.6	9.3	4.5	7.0
plus Avoided External Costs Energy (from AD, electricity) = C		11.5	19.2	7.9	14.3	4.9	11.6	4.5	9.3	7.6	15.5	4.8	10.8	10.4	16.1	11.8	19.3	7.0	14.1	7.1	13.1	5.0	10.8
plus Avoided External Costs Energy (from AD, CHP) = D		11.9	21.4	8.4	16.2	5.5	14.2	4.7	10.7	8.6	18.5	5.3	12.8	10.7	17.2	12.6	21.9	7.6	16.5	7.3	14.8	5.2	12.4
plus Landfill Disamenity = E		13.9	25.5	15.5	30.5	8.4	20.2	5.1	11.4	10.8	22.9	10.7	23.5	12.1	19.9	13.5	23.6	11.3	23.8	10.6	21.5	13.7	29.4
plus Avoided Private Costs Fertiliser = F		14.7	27.4	16.3	32.4	9.2	22.1	5.9	13.3	11.5	24.9	11.4	25.4	12.8	21.9	14.2	25.5	12.0	25.7	11.4	23.4	14.4	31.3
plus Avoided Private Costs of Pesticides = G		15.1	27.9	17.8	33.9	9.4	22.3	6.0	13.4	12.1	25.5	11.7	25.7	13.3	22.3	16.0	27.3	13.0	26.7	11.9	23.8	15.9	32.8
Private Costs																							
Landfill = H	Low	55.0	55.0	52.0	52.0	44.0	44.0	28.0	28.0	31.0	31.0	35.0	35.0	5.0	5.0	34.0	34.0	50.0	50.0	123.0	123.0	75.0	75.0
	High	110.0	110.0	56.0	56.0	44.0	44.0	56.0	56.0	85.0	85.0	220.0	220.0	21.0	21.0	78.0	78.0	70.0	70.0	147.0	147.0	75.0	75.0
AD plus Separate Collection = I	Low	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0
	High	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0
Net Private Cost = (I – H) = J	Low	25.0	25.0	28.0	28.0	36.0	36.0	52.0	52.0	49.0	49.0	45.0	45.0	75.0	75.0	46.0	46.0	30.0	30.0	-43.0	-43.0	5.0	5.0
	High	15.0	15.0	69.0	69.0	81.0	81.0	69.0	69.0	40.0	40.0	-95.0	-95.0	104.0	104.0	47.0	47.0	55.0	55.0	-22.0	-22.0	50.0	50.0
Net Private Cost Including Tax = J - landfill tax = K	Low	-18.0	-18.0	-24.0	-24.0	-14.0	-14.0	37.0	37.0	40.0	40.0	45.0	45.0	75.0	75.0	24.0	24.0	20.0	20.0	-43.0	-43.0	-55.0	-55.0
	High	-28.0	-28.0	14.0	17.0	31.0	31.0	54.0	54.0	31.0	31.0	-95.0	-95.0	104.0	104.0	25.0	25.0	5.0	5.0	-22.0	-22.0	-10.0	-10.0
Total Costs Net of External Benefits																							
Using Raw Score = K-A	Low	-28.5	-32.2	-31.3	-34.7	-18.2	-21.1	32.8	30.0	33.8	30.4	40.7	37.7	65.2	61.5	13.9	10.2	13.9	10.4	-49.8	-52.9	-59.8	-62.9
	High	-38.5	-42.2	6.7	6.3	26.8	23.9	49.8	47.0	24.8	21.4	-99.3	-102.3	94.2	90.5	14.9	11.2	-1.1	-4.6	-28.8	-31.9	-14.8	-17.9
Including Avoided External Costs of Energy (AD electricity) = K-B	Low	-29.5	-37.2	-31.9	-38.3	-18.9	-25.6	32.5	27.7	32.4	24.5	40.2	34.2	64.6	58.9	12.2	4.7	13.0	5.9	-50.1	-56.1	-60.0	-65.8
	High	-39.5	-47.2	6.1	2.7	26.1	19.4	49.5	44.7	23.4	15.5	-99.8	-105.8	93.6	87.9	13.2	5.7	-2.0	-9.1	-29.1	-35.1	-15.0	-20.8
Including Avoided External Costs of Energy (AD CHP) = K-C	Low	-29.9	-39.4	-32.4	-40.2	-19.5	-28.2	32.3	26.3	31.4	21.5	39.7	32.2	64.3	57.8	11.4	2.1	12.4	3.5	-50.3	-57.8	-60.2	-67.4
	High	-39.9	-49.4	5.6	0.8	25.5	16.8	49.3	43.3	22.4	12.5	-100.3	-107.8	93.3	86.8	12.4	3.1	-2.6	-11.5	-29.3	-36.8	-15.2	-22.4

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																					
Raw Score (direct emissions only) = A		8.9	12.5	8.3	11.9	4.2	7.0	4.5	7.5	11.1	14.9	11.1	14.9	10.6	14.1	9.9	13.6	11.1	14.9	10.0	13.7
plus Avoided External Costs Energy (from landfill = B)		8.8	12.2	8.1	11.5	3.9	6.3	3.9	6.5	11.1	14.8	11.1	14.8	10.5	14.0	9.8	13.5	11.0	14.8	9.9	13.6
plus Avoided External Costs Energy (from AD, electricity) = C		9.6	16.4	9.1	15.9	4.5	9.3	4.9	10.8	12.1	19.3	12.1	19.3	11.6	18.5	10.9	18.0	12.1	19.3	10.9	18.1
plus Avoided External Costs Energy (from AD, CHP) = D		9.9	18.2	9.5	17.8	4.7	10.5	5.4	12.7	12.5	21.3	12.5	21.3	12.0	20.5	11.3	19.9	12.5	21.2	11.4	20.0
plus Landfill Disamenity = E		11.7	21.9	10.8	20.3	5.3	11.7	10.6	23.2	13.8	23.8	14.6	25.3	12.5	21.5	13.0	23.2	14.5	25.1	12.9	23.0
plus Avoided Private Costs Fertiliser = F		12.5	23.9	11.6	22.3	6.0	13.6	11.4	25.1	14.6	25.7	15.3	27.2	13.3	23.4	13.7	25.2	15.2	27.0	13.7	25.0
plus Avoided Private Costs of Pesticides = G		13.1	24.5	11.8	22.5	6.2	13.7	12.0	25.8	16.0	27.2	16.8	28.7	13.5	23.6	14.2	25.6	16.7	28.5	14.3	25.6
Private Costs																					
Landfill = H	Low	6.0	6.0	6.0	6.0	22.0	22.0	10.0	10.0	5.0	5.0	15.0	15.0	15.0	15.0	15.0	15.0	15.0	15.0	5.0	5.0
	High	15.0	15.0	30.0	30.0	82.0	82.0	34.0	34.0	21.0	21.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	5.0	5.0
AD plus Separate Collection = I	Low	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0
	High	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0
Net Private Cost = (I – H) = J	Low	74.0	74.0	74.0	74.0	58.0	58.0	70.0	70.0	75.0	75.0	65.0	65.0	65.0	65.0	65.0	65.0	65.0	65.0	75.0	75.0
	High	110.0	110.0	95.0	95.0	43.0	43.0	91.0	91.0	104.0	104.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	120.0	120.0
Net Private Cost Including Tax = J - landfill tax = K	Low	74.0	74.0	74.0	74.0	28.0	28.0	51.0	51.0	75.0	75.0	64.0	64.0	65.0	65.0	65.0	65.0	65.0	65.0	75.0	75.0
	High	110.0	110.0	95.0	95.0	13.0	13.0	72.0	72.0	104.0	104.0	99.0	99.0	100.0	100.0	100.0	100.0	100.0	100.0	120.0	120.0
Total Costs Net of External Benefits																					
Using Raw Score = K-A	Low	65.1	61.5	65.7	62.1	23.8	21.0	46.5	43.5	63.9	60.1	52.9	49.1	54.4	50.9	55.1	51.4	53.9	50.1	65.0	61.3
	High	101.1	97.5	86.7	83.1	8.8	6.0	67.5	64.5	92.9	89.1	87.9	84.1	89.4	85.9	90.1	86.4	88.9	85.1	110.0	106.3
Including Avoided External Costs of Energy (AD electricity) = K-B	Low	64.4	57.6	64.9	58.1	23.5	18.7	46.1	40.2	62.9	55.7	51.9	44.7	53.4	46.5	54.1	47.0	52.9	45.7	64.1	56.9
	High	100.4	93.6	85.9	79.1	8.5	3.7	67.1	61.2	91.9	84.7	86.9	79.7	88.4	81.5	89.1	82.0	87.9	80.7	109.1	101.9
Including Avoided External Costs of Energy (AD CHP) = K-C	Low	64.1	55.8	64.5	56.2	23.3	17.5	45.6	38.3	62.5	53.7	51.5	42.7	53.0	44.5	53.7	45.1	52.5	43.8	63.6	55.0
	High	100.1	91.8	85.5	77.2	8.3	2.5	66.6	59.3	91.5	82.7	86.5	77.7	88.0	79.5	88.7	80.1	87.5	78.8	108.6	100.0

Table 43. Net External Costs And Private Costs Of Change From Landfill To AD (€ / tonne of waste switched, current landfill practice)

		AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																							
Raw Score (direct emissions only) = A		2.0	4.8	2.5	5.4	1.9	4.5	1.8	4.4	1.4	4.4	1.9	4.7	1.1	4.0	2.5	5.5	1.4	4.4	2.0	4.7	2.4	5.3
plus Avoided External Costs Energy (from landfill = B)		1.3	3.1	1.8	4.0	0.9	2.6	1.4	3.4	-0.2	2.1	1.2	3.2	0.7	3.2	1.1	3.5	0.4	2.6	1.7	3.5	2.1	4.1
tplus Avoided External Costs Energy (from AD, electricity) = C		2.3	8.3	2.8	8.4	2.3	8.6	2.0	6.5	2.0	9.1	2.3	7.8	1.3	5.8	3.0	9.4	1.8	8.0	2.1	7.3	2.5	7.9
plus Avoided External Costs Energy (from AD, CHP) = D		2.8	10.6	3.2	10.3	2.9	11.2	2.3	7.9	3.0	12.1	2.7	9.8	1.6	6.9	3.8	12.0	2.4	10.4	2.3	8.9	2.7	9.5
plus Landfill Disamenity = E		4.8	14.7	10.4	24.5	5.8	17.1	2.6	8.6	5.2	16.5	8.1	20.5	3.0	9.6	4.7	13.7	6.0	17.7	5.7	15.7	11.2	26.5
plus Avoided Private Costs Fertiliser = F		5.6	16.6	11.1	26.4	6.6	19.0	3.4	10.5	6.0	18.5	8.8	22.4	3.7	11.6	5.5	15.6	6.8	19.6	6.5	17.6	12.0	28.4
plus Avoided Private Costs of Pesticides = G		6.0	17.0	12.6	27.9	6.8	19.2	3.5	10.6	6.6	19.1	9.1	22.7	4.2	12.0	7.2	17.4	7.8	20.6	6.9	18.0	13.4	29.8
Private Costs																							
Landfill = H	Low	55.0	55.0	55.0	52.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	123.0	123.0	75.0	75.0
	High	110.0	110.0	56.0	56.0	55.0	55.0	56.0	56.0	85.0	85.0	220.0	220.0	55.0	55.0	78.0	78.0	70.0	70.0	147.0	147.0	75.0	75.0
AD plus Separate Collection = I	Low	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0
	High	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0
Net Private Cost = (I – H) = J	Low	25.0	25.0	25.0	28.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	-43.0	-43.0	5.0	5.0
	High	15.0	15.0	69.0	69.0	70.0	70.0	69.0	69.0	40.0	40.0	-95.0	-95.0	70.0	70.0	47.0	47.0	55.0	55.0	-22.0	-22.0	50.0	50.0
Net Private Cost Including Tax = J - landfill tax = K	Low	-18.0	-18.0	-27.0	-24.0	-25.0	-25.0	10.0	10.0	16.0	16.0	25.0	25.0	25.0	25.0	3.0	3.0	15.0	15.0	-43.0	-43.0	-55.0	-55.0
	High	-28.0	-28.0	14.0	17.0	20.0	20.0	54.0	54.0	31.0	31.0	-95.0	-95.0	70.0	70.0	25.0	25.0	5.0	5.0	-22.0	-22.0	-10.0	-10.0
Total Costs Net of External Benefits																							
Using Raw Score = K-A	Low	-20.0	-22.8	-29.5	-29.4	-26.9	-29.5	8.2	5.6	14.6	11.6	23.1	20.3	23.9	21.0	0.5	-2.5	13.6	10.6	-45.0	-47.7	-57.4	-60.3
	High	-30.0	-32.8	11.5	11.6	18.1	15.5	52.2	49.6	29.6	26.6	-96.9	-99.7	68.9	66.0	22.5	19.5	3.6	0.6	-24.0	-26.7	-12.4	-15.3
Including Avoided External Costs of Energy (AD electricity) = K-B	Low	-20.3	-26.3	-29.8	-32.4	-27.3	-33.6	8.0	3.5	14.0	6.9	22.7	17.2	23.7	19.2	0.0	-6.4	13.2	7.0	-45.1	-50.3	-57.5	-62.9
	High	-30.3	-36.3	11.2	8.6	17.7	11.4	52.0	47.5	29.0	21.9	-97.3	-102.8	68.7	64.2	22.0	15.6	3.2	-3.0	-24.1	-29.3	-12.5	-17.9
Including Avoided External Costs of Energy (AD CHP) = K-C	Low	-20.8	-28.6	-30.2	-34.3	-27.9	-36.2	7.7	2.1	13.0	3.9	22.3	15.2	23.4	18.1	-0.8	-9.0	12.6	4.6	-45.3	-51.9	-57.7	-64.5
	High	-30.8	-38.6	10.8	6.7	17.1	8.8	51.7	46.1	28.0	18.9	-97.7	-104.8	68.4	63.1	21.2	13.0	2.6	-5.4	-24.3	-30.9	-12.7	-19.5

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																					
Raw Score (direct emissions only) = A		1.2	4.1	1.1	3.9	1.8	4.5	2.1	4.9	2.4	5.4	2.4	5.4	1.9	4.6	1.2	4.0	2.4	5.3	1.2	4.1
plus Avoided External Costs Energy (from landfill = B)		0.7	2.7	0.4	2.5	1.4	3.5	1.4	3.5	1.7	3.9	1.7	3.9	1.2	3.1	0.4	2.6	1.7	3.9	0.5	2.7
plus Avoided External Costs Energy (from AD, electricity) = C		1.5	6.9	1.4	6.9	2.0	6.4	2.4	7.9	2.7	8.4	2.7	8.4	2.2	7.6	1.5	7.1	2.7	8.4	1.5	7.2
plus Avoided External Costs Energy (from AD, CHP) = D		1.8	8.7	1.8	8.8	2.2	7.7	2.8	9.8	3.2	10.4	3.2	10.4	2.6	9.6	1.9	9.0	3.1	10.3	2.0	9.1
plus Landfill Disamenity = E		3.6	12.4	3.1	11.3	2.8	8.8	8.0	20.2	4.4	12.9	5.2	14.4	3.2	10.6	3.6	12.4	5.1	14.2	3.5	12.1
plus Avoided Private Costs Fertiliser = F		4.4	14.4	3.8	13.3	3.6	10.8	8.8	22.1	5.2	14.8	5.9	16.3	3.9	12.5	4.4	14.3	5.8	16.1	4.3	14.1
plus Avoided Private Costs of Pesticides = G		5.0	15.0	4.1	13.5	3.7	10.9	9.5	22.8	6.6	16.3	7.4	17.8	4.1	12.7	4.8	14.8	7.3	17.6	4.9	14.7
Private Costs																					
Landfill = H	Low	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0
	High	55.0	55.0	55.0	55.0	82.0	82.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0	55.0
AD plus Separate Collection = I	Low	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0
	High	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0
Net Private Cost = (I – H) = J	Low	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0
	High	70.0	70.0	70.0	70.0	43.0	43.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0
Net Private Cost Including Tax = J - landfill tax = K	Low	25.0	25.0	25.0	25.0	-5.0	-5.0	6.0	6.0	25.0	25.0	24.0	24.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0	25.0
	High	70.0	70.0	70.0	70.0	13.0	13.0	51.0	51.0	70.0	70.0	69.0	69.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0	70.0
Total Costs Net of External Benefits																					
Using Raw Score = K-A	Low	23.8	20.9	23.9	21.1	-6.8	-9.5	3.9	1.1	22.6	19.6	21.6	18.6	23.1	20.4	23.8	21.0	22.6	19.7	23.8	20.9
	High	68.8	65.9	68.9	66.1	11.2	8.5	48.9	46.1	67.6	64.6	66.6	63.6	68.1	65.4	68.8	66.0	67.6	64.7	68.8	65.9
Including Avoided External Costs of Energy (AD electricity) = K-B	Low	23.5	18.1	23.6	18.1	-7.0	-11.4	3.6	-1.9	22.3	16.6	21.3	15.6	22.8	17.4	23.5	17.9	22.3	16.6	23.5	17.8
	High	68.5	63.1	68.6	63.1	11.0	6.6	48.6	43.1	67.3	61.6	66.3	60.6	67.8	62.4	68.5	62.9	67.3	61.6	68.5	62.8
Including Avoided External Costs of Energy (AD CHP) = K-C	Low	23.2	16.3	23.2	16.2	-7.2	-12.7	3.2	-3.8	21.8	14.6	20.8	13.6	22.4	15.4	23.1	16.0	21.9	14.7	23.0	15.9
	High	68.2	61.3	68.2	61.2	10.8	5.3	48.2	41.2	66.8	59.6	65.8	58.6	67.4	60.4	68.1	61.0	66.9	59.7	68.0	60.9

6.3.3 Costs Net of External Benefits

In the case of this switch, the costs net of benefits are more often positive (representing a positive cost to the change). This is the case for, in the low cost case, Finland, France, Greece, Italy, Portugal, Spain, UK and the Accession States. In the high cost case, the costs net of benefits are only negative in Austria, Germany, Luxembourg and the Netherlands.

6.4 Switching from Incineration to Composting

Table 44 below shows the external benefits, the private costs and the total economic cost of the switch from landfill to incineration. Incineration taxes are shown where they exist to illustrate how these would affect the calculus were they also to be included.

6.4.1 External Benefits

The external benefits in this case are rather larger than in the switch from landfill. Certainly, only where the incinerators are CHP plants, and under specific assumptions outlined earlier concerning the attribution of benefits to CHP incineration on the basis of 'avoided burdens', are there net disbenefits associated with switching to composting from incineration. Furthermore, this is only true in certain countries (Denmark, France and Italy). These are the countries in which the gas-fired power plants assessed in the ExternE study are the worst performers (in terms of their external costs) as assessed in the ExternE national implementation studies (European Commission 1999a). To put it another way, if these energy sources were cleaner, the benefits of a switch to composting become even less controversial than they already are.

This serves to underpin the way in which the benefits analysis hinges upon assumptions about 'avoided burdens' regarding energy generation, especially where the cases examined are energy from waste plants. The attribution of benefits to 'energy generated' from combustion of combustible wastes is likely to be especially controversial since some experts suggest that the low calorific value of these wastes makes it possible that if they were to be combusted on their own, they would not generate 'net energy'. The analysis is somewhat simplified in this regard since it simply uses calorific values of waste to derive net energy output without considering thresholds at which 'net energy' might be delivered.

Furthermore, it should be recalled that the costs of incineration often incorporate revenues for energy sales which are supported by energy recovery policies in the countries concerned. Therefore, to the extent that these enhancements for the price of energy delivered are in recognition of 'avoided burdens', potentially, the avoided burdens are being double-counted through their inclusion in the private and the external cost analysis.

Table 44. Net External Costs And Private Costs Of Change From Incineration To Composting (€ / tonne of waste switched)

		AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																							
Raw Score (direct emissions only) = A		14.3	23.4	15.2	23.1	12.1	19.8	12.1	20.3	11.9	24.1	12.5	23.3	11.5	22.1	11.5	22.2	12.0	21.9	13.9	20.4	14.0	21.9
Raw score plus incinerator disamenity = B		33.1	117.2	46.5	273.1	34.0	144.8	24.6	95.3	30.7	74.1	43.8	242.1	30.3	65.8	30.3	65.9	24.5	65.7	38.9	114.1	32.7	78.1
Raw score plus avoided burdens (energy production)																							
Energy as electricity only = C		10.2	13.8	11.1	14.9	6.5	8.7	9.5	14.6	3.0	11.1	8.1	14.8	8.9	17.2	8.9	17.4	6.5	11.9	12.1	13.3	12.1	14.8
Energy as CHP = D		5.0	1.4	5.9	4.4	-0.6	-5.7	6.1	7.2	-8.4	-5.6	2.3	3.8	5.6	11.0	5.6	11.2	-0.7	-1.0	9.7	4.2	9.7	5.8
Raw score plus avoided burdens and incinerator disamenity																							
Energy as electricity only = (C + B – A) = E		29.0	107.5	42.4	264.9	28.4	133.7	22.0	89.6	21.8	61.1	39.3	233.5	27.7	61.0	27.7	61.1	19.0	55.6	37.1	107.1	30.9	71.1
Energy as CHP = (D + B – A) = F		23.7	95.1	37.1	254.4	21.2	119.3	18.6	82.2	10.3	44.4	33.6	222.6	24.3	54.8	24.3	54.9	11.8	42.7	34.7	98.0	28.5	62.0
Private Costs																							
Incinerator = G	Low	95.0	95.0	84.0	84.0	43.0	43.0	70.0	70.0	69.0	69.0	90.0	90.0	70.0	70.0	70.0	70.0	100.0	100.0	97.0	97.0	90.0	90.0
	High	160.0	160.0	84.0	84.0	43.0	43.0	100.0	100.0	129.0	129.0	250.0	250.0	100.0	100.0	100.0	100.0	200.0	200.0	121.0	121.0	109.0	109.0
Compost plus Separate Collection = H	Low	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0
	High	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0
Net Private Cost = (H – G) = I	Low	-60.0	-60.0	-49.0	-49.0	-8.0	-8.0	-35.0	-35.0	-34.0	-34.0	-55.0	-55.0	-35.0	-35.0	-35.0	-35.0	-65.0	-65.0	-62.0	-62.0	-55.0	-55.0
	High	-85.0	-85.0	-9.0	-9.0	32.0	32.0	-25.0	-25.0	-54.0	-54.0	-175.0	-175.0	-25.0	-25.0	-25.0	-25.0	-125.0	-125.0	-46.0	-46.0	-34.0	-34.0
Total Costs Net of External Benefits																							
Using Raw Score = (I – A) = J	Low	-74.3	-83.4	-64.2	-72.1	-20.1	-27.8	-47.1	-55.3	-45.9	-58.1	-67.5	-78.3	-46.5	-57.1	-46.5	-57.2	-77.0	-86.9	-75.9	-82.4	-69.0	-76.9
	High	-99.3	-108.4	-24.2	-32.1	19.9	12.2	-37.1	-45.3	-65.9	-78.1	-187.5	-198.3	-36.5	-47.1	-36.5	-47.2	-137.0	-146.9	-59.9	-66.4	-48.0	-55.9
Raw score plus avoided burdens (electricity) = (I – C) = K	Low	-70.2	-73.8	-60.1	-63.9	-14.5	-16.7	-44.5	-49.6	-37.0	-45.1	-63.1	-69.8	-43.9	-52.2	-43.9	-52.4	-71.5	-76.9	-74.1	-75.3	-67.1	-69.8
	High	-95.2	-98.8	-20.1	-23.9	25.5	23.3	-34.5	-39.6	-57.0	-65.1	-183.1	-189.8	-33.9	-42.2	-33.9	-42.4	-131.5	-136.9	-58.1	-59.3	-46.1	-48.8
Raw score plus avoided burdens (CHP) = (I – D) = L	Low	-65.0	-61.4	-54.9	-53.4	-7.4	-2.3	-41.1	-42.2	-25.6	-28.4	-57.3	-58.8	-40.6	-46.0	-40.6	-46.2	-64.3	-64.0	-71.7	-66.2	-64.7	-60.8
	High	-90.0	-86.4	-14.9	-13.4	32.6	37.7	-31.1	-32.2	-45.6	-48.4	-177.3	-178.8	-30.6	-36.0	-30.6	-36.2	-124.3	-124.0	-55.7	-50.2	-43.7	-39.8
Memorandum Item - Incineration Tax	Low			6.0	6.0	37.0	37.0																
	High			13.0	13.0	44.0	44.0																

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
External Benefits																					
Raw Score (direct emissions only) = A		13.0	24.1	12.5	25.2	13.0	23.4	14.1	26.0	14.2	26.3	14.2	26.8	13.7	26.6	13.0	25.7	14.2	27.4	13.0	25.4
Raw score plus incinerator disamenity = B		22.4	39.7	37.5	181.5	31.8	73.4	45.3	244.8	26.7	70.1	32.9	120.5	26.2	57.8	31.7	119.4	32.9	121.1	25.5	37.9
Raw score plus avoided burdens (energy production)																					
Energy as electricity only = C		10.1	16.3	8.4	17.1	10.7	18.0	10.0	17.8	10.0	17.9	10.0	18.4	9.5	18.2	8.8	17.3	10.0	19.0	8.9	17.0
Energy as CHP = D		6.3	6.2	3.2	6.5	7.7	11.0	4.8	7.3	4.6	7.1	4.6	7.6	4.1	7.4	3.4	6.5	4.6	8.2	3.5	6.2
Raw score plus avoided burdens and incinerator disamenity																					
Energy as electricity only = (C + B – A) = E		19.4	31.9	33.4	173.3	29.4	68.0	41.3	236.6	22.5	61.7	28.7	112.2	22.0	49.5	27.5	111.1	28.8	112.7	21.4	29.5
Energy as CHP = (D + B – A) = F		15.6	21.9	28.2	162.8	26.4	61.0	36.0	226.1	17.1	50.9	23.4	101.4	16.6	38.7	22.2	100.3	23.4	102.0	16.0	18.7
Private Costs																					
Incinerator = G	Low	70.0	70.0	18.0	18.0	37.0	37.0	41.0	41.0	70.0	70.0	25.0	25.0	70.0	70.0	70.0	70.0	60.0	60.0	70.0	70.0
	High	100.0	100.0	51.0	51.0	87.0	87.0	66.0	66.0	100.0	100.0	40.0	40.0	100.0	100.0	100.0	100.0	60.0	60.0	100.0	100.0
Compost plus Separate Collection = H	Low	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0	35.0
	High	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0	75.0
Net Private Cost = (H – G) = I	Low	-35.0	-35.0	17.0	17.0	-2.0	-2.0	-6.0	-6.0	-35.0	-35.0	10.0	10.0	-35.0	-35.0	-35.0	-35.0	-25.0	-25.0	-35.0	-35.0
	High	-25.0	-25.0	24.0	24.0	-12.0	-12.0	9.0	9.0	-25.0	-25.0	35.0	35.0	-25.0	-25.0	-25.0	-25.0	15.0	15.0	-25.0	-25.0
Total Costs Net of External Benefits																					
Using Raw Score = (I – A) = J	Low	-48.0	-59.1	4.5	-8.2	-15.0	-25.4	-20.1	-32.0	-49.2	-61.3	-4.2	-16.8	-48.7	-61.6	-48.0	-60.7	-39.2	-52.4	-48.0	-60.4
	High	-38.0	-49.1	11.5	-1.2	-25.0	-35.4	-5.1	-17.0	-39.2	-51.3	20.8	8.2	-38.7	-51.6	-38.0	-50.7	0.8	-12.4	-38.0	-50.4
Raw score plus avoided burdens (electricity) = (I – C) = K	Low	-45.1	-51.3	8.6	-0.1	-12.7	-20.0	-16.0	-23.8	-45.0	-52.9	0.0	-8.4	-44.5	-53.2	-43.8	-52.3	-35.0	-44.0	-43.9	-52.0
	High	-35.1	-41.3	15.6	6.9	-22.7	-30.0	-1.0	-8.8	-35.0	-42.9	25.0	16.6	-34.5	-43.2	-33.8	-42.3	5.0	-4.0	-33.9	-42.0
Raw score plus avoided burdens (CHP) = (I – D) = L	Low	-41.3	-41.2	13.8	10.5	-9.7	-13.0	-10.8	-13.3	-39.6	-42.1	5.4	2.4	-39.1	-42.4	-38.4	-41.5	-29.6	-33.2	-38.5	-41.2
	High	-31.3	-31.2	20.8	17.5	-19.7	-23.0	4.2	1.7	-29.6	-32.1	30.4	27.4	-29.1	-32.4	-28.4	-31.5	10.4	6.8	-28.5	-31.2
Memorandum Item - Incineration Tax	Low																				
	High																				

Note that incinerator disamenity, as assessed here (through an extension of the hedonic pricing meta-analysis used by COWI (2000)), serves to swing the benefits equation massively in favour of composting. However, it must be recalled that a) this analysis is highly suspect (see Appendix 3) and b) there is no measurement here of disamenity arising from compost plants. These would certainly be expected to be non-zero, but to the extent that compost plants might be located in less densely populated areas (and they are not always), the disamenity as measured through hedonic pricing methods would be expected to be much lower than in the case of incineration (irrespective of the absolute magnitude of these). The estimates of incinerator disamenity canot be relied upon.

6.4.2 Private Costs

In most countries, the costs of treating waste through incineration are significantly in excess of the costs of treating waste through composting. As with landfill, however, the variation is large. Indeed, it is a feature of European waste management that gate fees for the different treatments still exhibit so much variation despite the effects of legislation which would appear to be harmonising the regulation of these treatments.

6.4.3 Costs Net of External Benefits

In almost all cases, the costs net of external benefits are negative. This suggests that costs outweigh benefits. The exceptions are in Denmark, Spain, the UK (CHP only) and the Czech Republic. In Denmark, the costs of incineration exceed the costs of source separation and composting, but the differential is less than the magnitude of the Danish incineration tax. In Spain and the Czech Republic, the low gate fees for incineration are the reason. In the UK, the gate fees are relatively low, but it is only for CHP plant that the net external benefits are less than private costs. It is worth noting that only one facility in the UK runs as a CHP plant.

6.4.4 Comment on Incinerator Taxes

Perhaps the most obvious comment that springs to mind is that the justification for implementing taxes on incineration appears at least as strong as that that which exists for taxing landfill. It is surprising, therefore, that more countries do not employ this instrument as a means to stimulate further source separation not only of compostables, but also of dry recyclables.

One explanation is that some countries have a more favourable view of the technology than others, so have tended to reduce gate fees through a range of explicit and implicit subsidies. Others, on the other hand, have regulated the technology more tightly, with the net effect that costs of running plant are higher in the first place. Another explanation that can be offered is that the ‘market’ for waste treatment becomes more competitive in countries which already achieve high rates of recycling and composting. In this context, issues of potential over-capacity in incineration start to become more prominent, so that a tax may have counter-productive effects.

6.5 Switching from Incineration to Anaerobic Digestion

The equivalent results for the switch from incineration to AD are shown in Table 45. The patterns are already well-established from the previous Tables.

6.5.1 External Benefits

This is the switch for which the external benefits are greatest. The benefits are greatest for the raw scores (i.e. with no ‘credits’ attributed for energy generation). Once avoided burdens for energy generation are introduced into the equation, the external benefits start to decline since the energy generated from AD is less than that from incineration in this analysis. Note that it was assumed that both plants either generate electricity only, or CHP. If the AD plant generated CHP and the incinerator delivered electricity only, the net external benefit would be increased (and vice versa).

6.5.2 Private Costs

The net private costs of this switch are negative in some countries and positive in others. The comments made concerning the earlier switches explain this.

6.5.3 Costs Net of External Benefits

Although this is the switch for which the external benefits are greatest, the costs net of external benefits are less favourable than in the case of the switch from composting to incineration. This is once again an illustration of the sensitivity of the net figure to the variation in private costs across countries.

6.6 Summary

The analysis confirms that, subject to the many limitations of the analysis, there appear to be external benefits associated with the separate collection of biowastes for composting / anaerobic digestion. The magnitude of these benefits is generally higher for the anaerobic digestion case where one assumes that the energy generated displaces burdens associated with energy generation from alternative sources (so that the ‘best case’ is that where anaerobic digestion is designed to deliver CHP). This is broadly consistent with other studies.

At the same time, in this analysis, the costs of AD and those of composting are assumed to be quite different. In this analysis, the additional costs of AD do not appear to justify the additional benefits. This conclusion should not be taken as a ‘strong’ conclusion, but it does suggest that those studies which suggest that merely because AD generates energy, it is superior to composting, have to be examined more closely. In particular, one comes back to the question of how it is that one treats the energy generated in terms of any external benefits assumed to be derived. If the underlying assumptions held that these were much higher (for example, where one assumes that the displaced source is the most polluting coal-fired power station), the additional benefits might justify the costs. Furthermore, were one to extend this assumption to incineration plants, these would then appear to perform more favourably.

Table 45. Net External Costs And Private Costs Of Change From Incineration To AD (€ / tonne of waste switched)

		AU		BE		DE		FI		FR		GE		GR		IR		IT
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low
External Benefits																		
Raw Score (direct emissions only) = A		15.5	27.4	16.9	27.2	13.1	23.2	12.4	22.7	13.9	28.9	13.7	26.9	12.1	25.0	13.4	26.4	13.4
Raw score plus incinerator disamenity = B		34.2	121.1	48.2	277.2	34.9	148.2	24.9	97.7	32.6	78.9	44.9	245.7	30.8	68.7	32.2	70.2	25.9
Raw score plus avoided burdens (energy production)																		
Incinerator and AD With Electricity only = C		12.4	22.9	13.8	23.4	8.9	18.1	10.5	20.0	7.2	22.9	10.3	23.0	10.1	22.7	12.7	27.5	9.2
Incinerator and AD with CHP = D		7.6	12.7	9.0	14.8	2.3	6.3	7.4	14.0	-3.3	9.2	5.1	14.0	7.0	17.6	10.2	23.8	2.7
Raw score plus avoided burdens and incinerator disamenity																		
Incinerator and AD With Electricity only = (C + B – A) = E		31.1	116.7	45.1	273.4	30.8	143.1	23.0	95.0	25.9	72.9	41.6	241.7	28.9	66.5	31.4	71.2	21.7
Incinerator and AD with CHP = (D + B – A) = F		26.3	106.5	40.3	264.8	24.2	131.3	19.9	89.0	15.4	59.2	36.3	232.7	25.8	61.4	28.9	67.6	15.2
Private Costs																		
Incinerator = G	Low	95.0	95.0	84.0	84.0	43.0	43.0	70.0	70.0	69.0	69.0	90.0	90.0	70.0	70.0	70.0	70.0	100.0
	High	160.0	160.0	84.0	84.0	43.0	43.0	100.0	100.0	129.0	129.0	250.0	250.0	100.0	100.0	100.0	100.0	200.0
AD plus Separate Collection = H	Low	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0
	High	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0
Net Private Cost = (H – G) = I	Low	-15.0	-15.0	-4.0	-4.0	37.0	37.0	10.0	10.0	11.0	11.0	-10.0	-10.0	10.0	10.0	10.0	10.0	-20.0
	High	-35.0	-35.0	41.0	41.0	82.0	82.0	25.0	25.0	-4.0	-4.0	-125.0	-125.0	25.0	25.0	25.0	25.0	-75.0
Total Costs Net of External Benefits																		
Using Raw Score = (I – A) = J	Low	-30.5	-42.4	-20.9	-31.2	23.9	13.8	-2.4	-12.7	-2.9	-17.9	-23.7	-36.9	-2.1	-15.0	-3.4	-16.4	-33.4
	High	-50.5	-62.4	24.1	13.8	68.9	58.8	12.6	2.3	-17.9	-32.9	-138.7	-151.9	12.9	0.0	11.6	-1.4	-88.4
Raw score plus avoided burdens (electricity) = (I – C) = K	Low	-27.4	-37.9	-17.8	-27.4	28.1	18.9	-0.5	-10.0	3.8	-11.9	-20.3	-33.0	-0.1	-12.7	-2.7	-17.5	-29.2
	High	-47.4	-57.9	27.2	17.6	73.1	63.9	14.5	5.0	-11.2	-26.9	-135.3	-148.0	14.9	2.3	12.3	-2.5	-84.2
Raw score plus avoided burdens (CHP) = (I – D) = L	Low	-22.6	-27.7	-13.0	-18.8	34.7	30.7	2.6	-4.0	14.3	1.8	-15.1	-24.0	3.0	-7.6	-0.2	-13.8	-22.7
	High	-42.6	-47.7	32.0	26.2	79.7	75.7	17.6	11.0	-0.7	-13.2	-130.1	-139.0	18.0	7.4	14.8	1.2	-77.7
Memorandum Item - Incineration Tax	Low			6.0	6.0	37.0	37.0											
	High			13.0	13.0	44.0	44.0											

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low
External Benefits																		
Raw Score (direct emissions only) = A		13.1	24.2	12.7	25.6	12.8	23.1	14.3	26.3	14.4	26.7	14.4	27.2	13.9	27.0	13.2	26.1	14.4
Raw score plus incinerator disamenity = B		22.5	39.8	37.7	181.8	31.6	73.1	45.6	245.1	26.9	70.4	33.2	120.9	26.4	58.2	32.0	119.8	33.2
Raw score plus avoided burdens (energy production)																		
Incinerator and AD With Electricity only = C		10.9	20.6	9.6	21.8	11.1	20.6	11.2	22.5	11.3	22.8	11.3	23.3	10.8	23.1	10.1	22.2	11.3
Incinerator and AD with CHP = D		7.4	12.4	4.8	13.2	8.3	14.8	6.4	13.9	6.3	14.0	6.4	14.5	5.9	14.3	5.1	13.4	6.4
Raw score plus avoided burdens and incinerator disamenity																		
Incinerator and AD With Electricity only = (C + B – A) = E		20.2	36.2	34.6	178.1	29.8	70.6	42.5	241.3	23.8	66.6	30.0	117.1	23.3	54.4	28.8	115.9	30.0
Incinerator and AD with CHP = (D + B – A) = F		16.7	28.0	29.8	169.5	27.1	64.8	37.7	232.7	18.8	57.7	25.1	108.2	18.4	45.5	23.9	107.1	25.1
Private Costs																		
Incinerator = G	Low	70.0	70.0	18.0	18.0	37.0	37.0	41.0	41.0	70.0	70.0	25.0	25.0	70.0	70.0	70.0	70.0	60.0
	High	100.0	100.0	51.0	51.0	87.0	87.0	66.0	66.0	100.0	100.0	40.0	40.0	100.0	100.0	100.0	100.0	60.0
AD plus Separate Collection = H	Low	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0	80.0
	High	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0	125.0
Net Private Cost = (H – G) = I	Low	10.0	10.0	62.0	62.0	43.0	43.0	39.0	39.0	10.0	10.0	55.0	55.0	10.0	10.0	10.0	10.0	20.0
	High	25.0	25.0	74.0	74.0	38.0	38.0	59.0	59.0	25.0	25.0	85.0	85.0	25.0	25.0	25.0	25.0	65.0
Total Costs Net of External Benefits																		
Using Raw Score = (I – A) = J	Low	-3.1	-14.2	49.3	36.4	30.2	19.9	24.7	12.7	-4.4	-16.7	40.6	27.8	-3.9	-17.0	-3.2	-16.1	5.6
	High	11.9	0.8	61.3	48.4	25.2	14.9	44.7	32.7	10.6	-1.7	70.6	57.8	11.1	-2.0	11.8	-1.1	50.6
Raw score plus avoided burdens (electricity) = (I – C) = K	Low	-0.9	-10.6	52.4	40.2	31.9	22.4	27.8	16.5	-1.3	-12.8	43.7	31.7	-0.8	-13.1	-0.1	-12.2	8.7
	High	14.1	4.4	64.4	52.2	26.9	17.4	47.8	36.5	13.7	2.2	73.7	61.7	14.2	1.9	14.9	2.8	53.7
Raw score plus avoided burdens (CHP) = (I – D) = L	Low	2.6	-2.4	57.2	48.8	34.7	28.2	32.6	25.1	3.7	-4.0	48.6	40.5	4.1	-4.3	4.9	-3.4	13.6
	High	17.6	12.6	69.2	60.8	29.7	23.2	52.6	45.1	18.7	11.0	78.6	70.5	19.1	10.7	19.9	11.6	58.6
Memorandum Item - Incineration Tax	Low																	
	High																	

That having been said, the assumptions made regarding energy displacement seem more defensible than those of earlier studies (which have tended to assume displacement of the most polluting energy source). This is not really tenable in the context of analysis of the effects of policy decisions the effects of which are likely to be felt not only today, but ten to twenty years in the future. In this period, it is expected that electricity generation will shift more sharply towards renewable sources, and there will be continuing pressure applied to all power generation sources to reduce emissions of all pollutants (for example, through the IPPC Directive).

The fundamental impact that this assumption has on the results of the analysis cannot, however, be denied. This is effectively shown in the way in which the external costs change as one moves from the ‘raw scores’ to the scores generated assuming some avoided burdens associated with energy generation.

Another reason for the ‘weakness’ of this conclusion is that the private costs of anaerobic digestion are expected to fall. This will occur as understanding of the bacteriological processes improves, speeding up throughput of material, and enabling more material to be treated for a given cost.

Lastly, the continued variation in private costs of landfill and incineration across Europe is deserving of much closer examination. It seems clear that in those countries where source separation has been pursued most vigorously, landfill gate fees are relatively high. This may be either, or both, an explanatory variable in the shift towards more source separation, or the consequence of a wider policy aimed at encouraging the activity (for example, through mandating the public / local authorities, or establishing targets, as has happened in several EU countries / regions).

The variation in incineration gate fees probably reflects a range of implicit and explicit subsidies associated with capital financing, the generation of ‘renewable energy’, and other distorting instruments applied at the level of Member States (including differences in regulatory measures). Once again, however, with the exception of Denmark, the gate fees in countries where source separation is far advanced are relatively high. Denmark, however, has the highest incineration tax of any country in Europe (the world). Indeed, the tax alone is greater than the total gate fee paid in many countries such as Spain, the Czech Republic and the UK (in some cases).

These comments suggest that those countries where the waste management hierarchy is most closely followed are those where the structure of gate fees reflects that hierarchy. By implication, if the intention is to ensure other Member States do the same, the argument for ensuring still closer harmonisation in the legislation, regulation and incentives which deliver such varied gate fees seems a strong one. In particular, in those nations where measures have been introduced which reduce the costs of residual waste treatments, these should be examined for the potentially distorting and environmentally perverse incentives which they convey with respect to source separation for both recycling and composting / AD. Indeed, recyclers / composters may see these as being unjustified state aids to specific industries. The need to ensure incentive compatibility across instruments emanating from the ‘renewable energy’ domain and those related to waste policy seems particularly pertinent at the present moment.

7.0 FUTURE SCENARIOS

7.1 Baseline Scenario - Influence of the Landfill Directive

In order to understand how different policies might affect municipal waste in EU Member States, one has to understand the nature of the changes which such policies would imply. In this context, the 'baseline' scenario against which new policy interventions must be measured is a world in which the full effects of the Landfill Directive are taken into account. The issues arising in this context are discussed in more detail in Appendix 7.

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

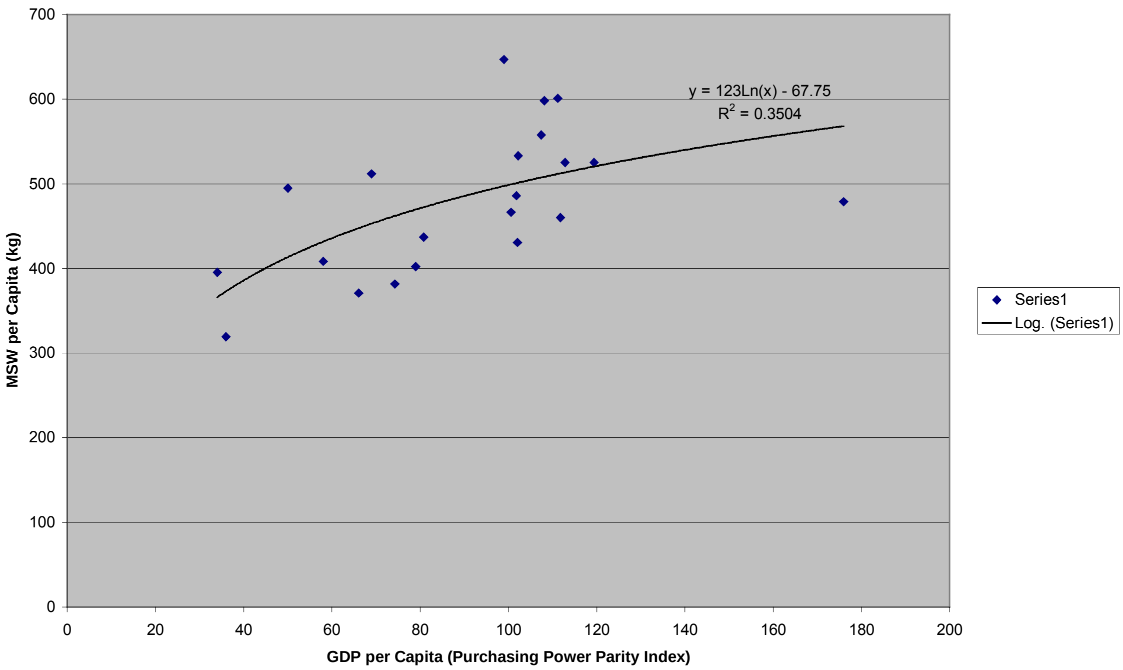
- Arisings (i.e. rates of growth in arisings);
- Composition and its evolution; and
- Likely future treatment options.

In the ideal world, the first two are well known, and the assignment of different waste fractions to specific treatment options follows a more or less well understood National Plan. In practice:

- growth rates are not easy to anticipate, especially over the twenty year period over which the Landfill Directive takes effect (Figure 14 seeks to illustrate some relationship between per capita MSW and purchasing power index). Projecting forward on the basis of recently quoted growth rates / aims in waste plans is an apparently futile exercise given the improbability of the resulting divergent quantities of waste generated when these are projected forward for longer periods of time (see Figure 15);
- composition, though it is bound to change, will change in ways that are difficult to estimate. It is not possible to predict such changes in advance an over extended period of time; and
- the future treatment options are not so well known over this period.

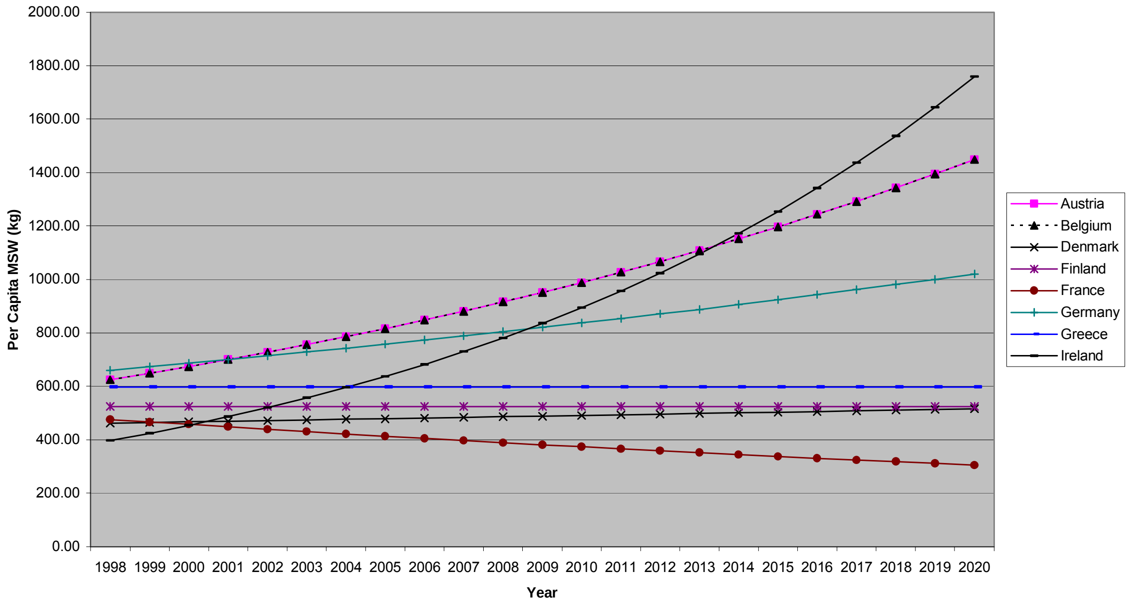
The following country classification is proposed for modelling purposes (see Appendix 7 for further details regarding projections). From the perspective of the model, this defines what treatment options are likely to be used to meet the Landfill Directive targets in the countries concerned:

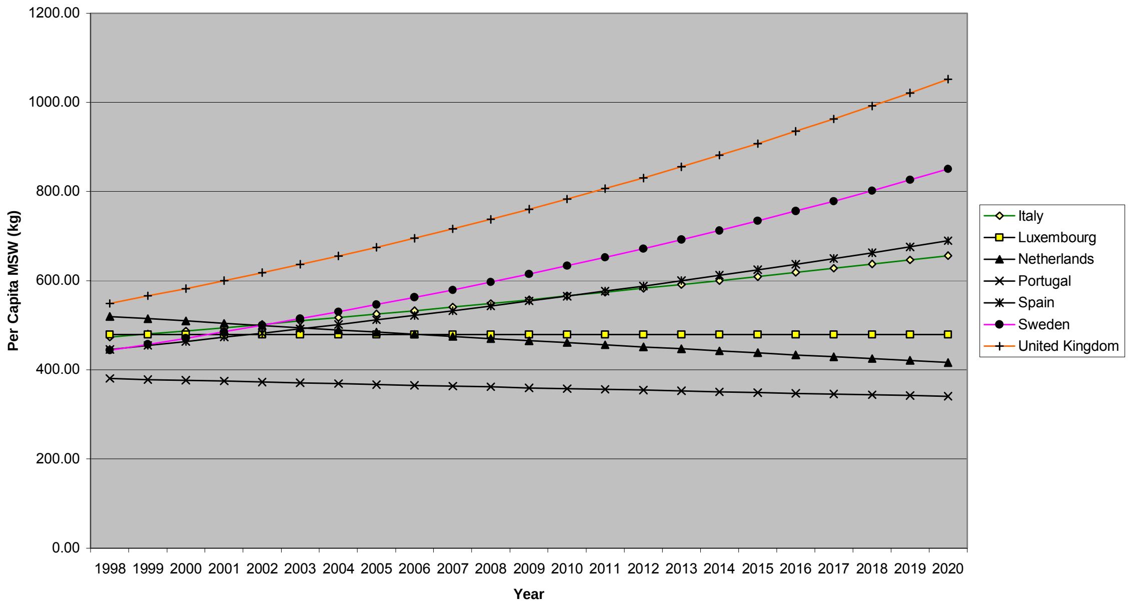
Figure 14: Plot Of Per Capita MSW (kg) .v. GDP per Capita (Purchasing Power Index)



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

Figure 15: Evolution of Per Capita MSW in Member States





- Group 2 - Finland, Belgium (Wallonia), Slovenia, Estonia, Sweden, Denmark, Spain, Italy, France, Poland, Wales (UK) and Scotland (UK) – in these countries, there is an intention to develop separate collection quite swiftly, but this is not especially widely diffused at present. In these countries, additional diversion of biodegradable wastes from landfill is expected to be achieved through increased recycling of paper (though in Denmark, Sweden and Finland, this is already well-advanced), more source separation of compostables, and incineration. It is anticipated that the switch from the Landfill Directive compliance trajectory to that required under a proposed Biowaste Directive would be principally one from incineration of unseparated biowastes to composting / anaerobic digestion of separately collected fractions, possibly with MBT used as source separation is developed (see below). This shift is unlikely to be extremely pronounced. In particular, Finland has set high targets for the recovery of both paper and kitchen and yard waste, with recovery understood to be principally recycling, composting and anaerobic digestion. Finland, therefore, could be a ‘Group 1’ country;
- Group 3: Ireland, England (UK), Northern Ireland (UK), Portugal, Belgium (Brussels), Czech Republic, Cyprus and Hungary. In these countries, relatively little separate collection of compostables is being undertaken at present. Landfill Directive targets likely to be met through a mix of incineration, paper recycling, and composting of yard waste. Some composting of source separated kitchen wastes may occur, but mixed waste composting and incineration are likely to be the most common routes for treatment of biowaste other than yard waste delivered to containerparks / civic amenity sites. The effect of a requirement for source separation would be a pronounced shift away from incineration / mixed waste composting towards composting / digestion of source separated waste and, possibly, MBT (used as pre-treatment whilst source separation develops – see below); and
- Group 4: Greece. Very little separate collection is being undertaken at present and there appear to be no major pushes for this to occur in future. Landfill Directive compliance is likely to be pursued through mixed waste composting and paper recycling. There may be some incineration but public support for this is absent. The proposed policy change would require radical changes

7.1.1 Treatments Under the Baseline

The approach to estimating how treatments might develop under the Landfill Directive follows partly from the analysis above. However, the quantity of biodegradable municipal waste required to be diverted from landfill changes in absolute terms depending upon the growth rate being assumed. This is the subject of considerable uncertainty over the years ahead, and there is reason to believe that countries which are taking measures to minimise growth in arisings will experience lower growth rates than others. Ranges of the order –1% - +3% seem reasonable to use over this period with those undertaking minimisation measures more likely to be at lower end of this range.

Attention is focussed on the fate of the kitchen and garden wastes under the Directive, though the fate of paper clearly influences the trajectory for compliance under the Landfill Directive. Effectively, it has been assumed that the Article 5 targets, which apply to biodegradable wastes as a whole, are being applied to the kitchen and garden wastes specifically. This is a tolerable approximation given all the other confounding factors which hinder this type of analysis.

Country groupings established above have been used to develop decision rules for the implications of the Landfill Directive for the remaining landfilled kitchen and garden wastes. These are:

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

For the purposes of this analysis, the whole of the UK is treated as a Group 3 country with Greece also in the same Group. The ‘baseline’ parameters characterising the situation posed by the Landfill Directive is shown in Table 46. Landfill diversion assumed to occur in the year 2010 for different growth rates is shown in Table 47.

Table 46. Baseline Parameters for Biowaste Going to Landfill Under Landfill Directive (tonnes)

	AU	BE	DK	FI	FR	GER	GRE	IRE	IT	LUX	NL
	1	1	2	2	3	1	3	3	2	1	1
Biowaste collected (latest)	594,793	800,000	637,000	93,000	970,000	7,000,000	0	6,000	1,500,000	34,000	1,790,000
Biowaste collected 1995 (estimate)	505,574	700,000	541,450	79,050	824,500	5,950,000	0	5,100	1,275,000	28,900	1,521,500
Total biowaste available	905,200	1,668,795	1,028,600	1,004,000	11,020,000	14,680,900	1,833,000	556,200	9,021,600	130,962	3,452,400
Biowaste remaining (1995)	912,086	968,795	487,150	924,950	10,195,500	8,730,900	1,833,000	551,100	7,746,600	102,062	1,930,900
Fraction biowaste landfilled (1995)	601,723	541,386	85,805	848,565	6,835,619	5,966,115	1,833,000	551,100	7,159,052	33,660	526,924
Allowed to Landfill (2010)	300,862	270,693	42,902	424,283	5,126,714	2,983,058	1,374,750	413,325	5,369,289	16,830	263,462
	POR	SP	SW	UK	CYP	CZ	EST	HUN	POL	SLO	
	3	3	2	3	3	2	2	2	2	2	
Biowaste collected (latest)	14,000	0	400,000	615,000	0	385,000	1,000	0	222,000	10,000	
Biowaste collected 1995 (estimate)	11,900	0	340,000	522,750	0	327,250	850	0	188,700	8,500	
Total biowaste available	1,406,000	7,585,200	1,524,000	12,001,500	132,258	1,466,500	301,570	1,875,000	4,680,460	330,752	
Biowaste remaining (1995)	1,394,100	7,585,200	1,184,000	11,478,750	132,258	1,139,250	300,720	1,875,000	4,491,760	322,252	
Fraction biowaste landfilled (1995)	1,394,100	7,020,141	491,060	11,478,750	121,685	1,139,250	300,720	1,757,813	4,446,389	322,252	
Allowed to Landfill (2010)	1,045,575	5,265,106	245,530	8,609,063	91,264	569,625	225,540	1,318,359	3,334,792	241,689	

Table 47. Estimated Baseline Treatments of Biowaste ('000 tonnes) Under the Landfill Directive at Different Growth Rates, 2010

			AU	BE	DK	FI	FR	GER	GRE	IRE	IT	LUX	NL
Total Biowaste	-1%		802	1,479	912	890	9,768	13,013	1,625	493	7,997	116	3,060
Growth	0%		905	1,669	1,029	1,004	11,020	14,681	1,833	556	9,022	131	3,452
Rates	1%		1,020	1,880	1,159	1,131	12,418	16,543	2,065	627	10,166	148	3,890
	2%		1,148	2,116	1,305	1,273	13,976	18,619	2,325	705	11,442	166	4,378
Apportioned Biowaste			0	0	0	0	0	0	0	0	0	0	0
	-1%	*SS*	642	1,183	547	534	2,930	10,410	487	148	4,798	93	2,448
		AD	64	118	55	53	293	1,041	49	15	480	9	245
		COMP	578	1,065	492	481	2,637	9,369	439	133	4,318	84	2,203
		REMAINDER	160	296	365	356	6,838	2,603	1,137	345	3,199	23	612
		INCIN	80	148	274	267	1,283	1,301	0	0	0	12	306
		MBT	80	148	91	89	428	1,301	0	0	0	12	306
		LANDFILL NO PRE-TREATMENT	0	0	0	0	5,127	0	1,137	345	3,199	0	0
			0	0	0	0	0	0	0	0	0	0	0
	0%	*SS*	724	1,335	617	602	3,306	11,745	550	167	5,413	105	2,762
		AD	72	134	62	60	331	1,174	55	17	541	10	276
		COMP	652	1,202	555	542	2,975	10,570	495	150	4,872	94	2,486
		REMAINDER	181	334	411	402	7,714	2,936	1,283	389	3,609	26	690
		INCIN	91	167	309	301	1,940	1,468	0	0	0	13	345
		MBT	91	167	103	100	647	1,468	0	0	0	13	345
		LANDFILL NO PRE-TREATMENT	0	0	0	0	5,127	0	1,283	389	3,609	0	0
			0	0	0	0	0	0	0	0	0	0	0
	1%	*SS*	816	1,504	695	679	3,725	13,234	620	188	6,099	118	3,112
		AD	82	150	70	68	373	1,323	62	19	610	12	311
		COMP	734	1,354	626	611	3,353	11,911	558	169	5,490	106	2,801
		REMAINDER	204	376	464	453	8,692	3,309	1,446	439	4,066	30	778
		INCIN	102	188	348	339	2,674	1,654	53	19	0	15	389
		MBT	102	188	116	113	891	1,654	18	6	0	15	389
		LANDFILL NO PRE-TREATMENT	0	0	0	0	5,127	0	1,375	413	4,066	0	0
			0	0	0	0	0	0	0	0	0	0	0
	2%	*SS*	918	1,693	783	764	4,193	14,895	697	212	6,865	133	3,503
		AD	92	169	78	76	419	1,490	70	21	686	13	350
		COMP	827	1,524	704	688	3,774	13,406	628	190	6,178	120	3,153
		REMAINDER	230	423	522	509	9,783	3,724	1,627	494	4,577	33	876
		INCIN	115	212	391	382	3,492	1,862	189	60	0	17	438
		MBT	115	212	130	127	1,164	1,862	63	20	0	17	438
		LANDFILL NO PRE-TREATMENT	0	0	0	0	5,127	0	1,375	413	4,577	0	0

			POR	SP	SW	UK	CYP	CZ	EST	HUN	POL	SLO
Total Biowaste												
Growth	-1%		1,246	6,723	1,351	10,638	117	1,300	267	1,662	4,149	293
Rates	0%		1,406	7,585	1,524	12,002	132	1,467	302	1,875	4,680	331
	1%		1,584	8,547	1,717	13,524	149	1,652	340	2,113	5,274	373
	2%		1,783	9,620	1,933	15,221	168	1,860	382	2,378	5,936	419
Apportioned Biowaste			0	0	0	0	0	0	0	0	0	0
	-1%	*SS*	374	2,017	811	3,191	35	390	160	499	2,489	176
		AD	37	202	81	319	4	39	16	50	249	18
		COMP	336	1,815	729	2,872	32	351	144	449	2,240	158
		REMAINDER	872	4,706	540	7,447	82	910	107	1,163	1,659	117
		INCIN	0	0	405	0	0	255	0	0	0	0
		MBT	0	0	135	0	0	85	0	0	0	0
		LANDFILL NO PRE-TREATMENT	872	4,706	0	7,447	82	570	107	1,163	1,659	117
			0	0	0	0	0	0	0	0	0	0
	0%	*SS*	422	2,276	914	3,600	40	440	181	563	2,808	198
		AD	42	228	91	360	4	44	18	56	281	20
		COMP	380	2,048	823	3,240	36	396	163	506	2,527	179
		REMAINDER	984	5,310	610	8,401	93	1,027	121	1,313	1,872	132
		INCIN	0	33	457	0	1	343	0	0	0	0
		MBT	0	11	152	0	0	114	0	0	0	0
		LANDFILL NO PRE-TREATMENT	984	5,265	0	8,401	91	570	121	1,313	1,872	132
			0	0	0	0	0	0	0	0	0	0
	1%	*SS*	475	2,564	1,030	4,057	45	496	204	634	3,164	224
		AD	48	256	103	406	4	50	20	63	316	22
		COMP	428	2,308	927	3,651	40	446	184	570	2,848	201
		REMAINDER	1,109	5,983	687	9,467	104	1,157	136	1,479	2,110	149
		INCIN	48	538	515	643	10	440	0	120	0	0
		MBT	16	179	172	214	3	147	0	40	0	0
		LANDFILL NO PRE-TREATMENT	1,046	5,265	0	8,609	91	570	136	1,318	2,110	149
			0	0	0	0	0	0	0	0	0	0
	2%	*SS*	535	2,886	1,160	4,566	50	558	229	713	3,562	252
		AD	53	289	116	457	5	56	23	71	356	25
		COMP	481	2,597	1,044	4,110	45	502	207	642	3,205	227
		REMAINDER	1,248	6,734	773	10,655	117	1,302	153	1,665	2,374	168
		INCIN	152	1,102	580	1,534	20	549	0	260	0	0
		MBT	51	367	193	511	7	183	0	87	0	0
		LANDFILL NO PRE-TREATMENT	1,046	5,265	0	8,609	91	570	153	1,318	2,374	168

7.2 Effects of Implementing a Requirement for Source Separation of Materials (as well as standards)

It is possible (though certainly not straightforward) to estimate the changes in the treatments which Member States might apply to the biowaste fraction of municipal waste if there was a requirement to collect separately all biodegradable wastes. A requirement to source separate municipal wastes would certainly lead to greater resort to compost and anaerobic digestion systems. This might also have a knock-on effects on approaches to waste collection and treatment. In particular, source separation of other materials might well be introduced more widely alongside separate collection of biowastes. Furthermore, the instatement of separate collection of biowastes probably makes the introduction of direct charging systems both more appropriate and more effective. Consequently, a positive indirect effect of implementing source separation schemes might be to encourage the reduction in waste materials set out for collection, both through enabling direct charging for wastes (without the otherwise-more-likely side-effect of increased fly-tipping) and through sensitising citizens to the nature of the wastes which they generate in their daily lives.

Most obviously, those countries which are likely to seek to divert greater proportions of biowaste from landfill (under the Landfill Directive) through incineration and MBT will be required to achieve more of their diversion through composting and anaerobic digestion. The degree to which this occurs as a consequence of the policy depends upon:

a) the degree to which this path would have been followed anyway (as a means to achieve Landfill Directive targets –estimates as to what will happen are outlined above); and

b) the maximum level of materials capture achievable through these systems.

The former is the subject of some conjecture. The latter is likely to be of the order 85% for all biowastes, though higher rates are reported for garden wastes individually (e.g. in Denmark, where 95% of garden wastes are collected for composting).

7.2.1 Effects Relative to Baseline – Quantity of Biowastes Separately Collected

The impacts of the proposed policy change, requiring separate collection, are estimated through modelling the capture rate for kitchen and yard wastes rising to a maximum of 85% in 2010 from households which are covered by the Directive. The estimated changes in quantities collected separately, and the implied switches from landfill and incineration (relative to a 'Landfill Directive only' scenario) are shown in Table 48. Note that these are changes for the specific year, not for a cumulative change which occurs over time.

The analysis focuses on shifts of kitchen and yard wastes between the four principal treatments examined in this model, landfill, incineration, composting and anaerobic digestion. This means it has been assumed that shifts are away from landfill and incineration. The wisdom of projecting forward through to the years beyond 2010 is taken to highly questionable. Already, the approach implies a considerable level of guess-work and interpolation (given the low quality of much of the data available and the uncertainties surrounding future waste management strategies). To pretend that the analysis being undertaken here has validity to the year 2020 is likely to mislead as much as it is to inform. Even the results to 2010 should be treated with considerable caution.

Table 48. Changes in Level of Source Separation and Switches Between Treatments for Biowaste Resulting From Implementation of a Requirement for Source Separation (tonnes biowaste in year 2010)

		AU	BE	DK	FI	FR	GER	GRE	IRE	IT	LUX	NL
Total												
	-1%	40,118	73,960	227,934	222,483	5,372,379	650,646	893,609	271,154	1,999,152	5,804	153,008
	0%	45,260	83,440	257,150	251,000	6,061,000	734,045	1,008,150	305,910	2,255,400	6,548	172,620
	1%	51,000	94,022	289,763	282,833	6,829,687	827,140	1,136,009	344,707	2,541,441	7,379	194,513
	2%	57,401	105,822	326,128	318,329	7,686,814	930,947	1,278,578	387,968	2,860,393	8,305	218,924
Switch from landfill												
	-1%	0	0	0	0	4,089,235	0	893,609	271,154	1,999,152	0	0
	0%	0	0	0	0	4,120,536	0	1,008,150	305,910	2,255,400	0	0
	1%	0	0	0	0	4,155,476	0	1,082,699	325,662	2,541,441	0	0
	2%	0	0	0	0	4,194,436	0	1,089,180	327,629	2,860,393	0	0
Switch from incineration												
	-1%	40,118	73,960	227,934	222,483	1,283,144	650,646	0	0	0	5,804	153,008
	0%	45,260	83,440	257,150	251,000	1,940,464	734,045	0	0	0	6,548	172,620
	1%	51,000	94,022	289,763	282,833	2,674,210	827,140	53,309	19,045	0	7,379	194,513
	2%	57,401	105,822	326,128	318,329	3,492,377	930,947	189,398	60,339	0	8,305	218,924
		POR	SP	SW	UK	CYP	CZ	EST	HUN	POL	SLO	
Total	-1%	685,441	3,697,874	337,713	5,850,871	64,477	714,936	66,827	914,084	1,037,172	73,293	
	0%	773,300	4,171,860	381,000	6,600,825	72,742	806,575	75,393	1,031,250	1,170,115	82,688	
	1%	871,374	4,700,956	429,320	7,437,975	81,967	908,869	84,954	1,162,038	1,318,515	93,175	
	2%	980,731	5,290,927	483,200	8,371,442	92,254	1,022,932	95,616	1,307,874	1,483,989	104,868	
Switch from landfill												
	-1%	685,441	3,697,874	0	5,850,871	64,477	459,716	66,827	914,084	1,037,172	73,293	
	0%	773,300	4,138,459	0	6,600,825	71,754	463,881	75,393	1,031,250	1,170,115	82,688	
	1%	823,789	4,162,509	0	6,794,887	72,174	468,531	84,954	1,041,589	1,318,515	93,175	
	2%	828,760	4,189,326	0	6,837,317	72,641	473,716	95,616	1,048,218	1,483,989	104,868	
Switch from incineration												
	-1%	0	0	337,713	0	0	255,220	0	0	0	0	
	0%	0	33,401	381,000	0	987	342,694	0	0	0	0	
	1%	47,585	538,447	429,320	643,088	9,794	440,338	0	120,449	0	0	
	2%	151,971	1,101,601	483,200	1,534,125	19,613	549,216	0	259,656	0	0	

7.2.2 Costs and Benefits of Implementing a Requirement for Source Separation

The external benefits of the changes being required under the requirement for separate collection are shown in Table 49. For the 'central estimates' of the external benefits of switching which are reported, the following assumptions were used:

- For landfill to compost: the figure which includes avoided external costs associated with energy generated by the landfill, but not landfill disamenity.

- For landfill to AD: the figure which includes avoided external costs associated with electricity generated by both, but not landfill disamenity;

- For incineration to compost: the figure which includes avoided external costs associated with electricity from the incinerator, but not incinerator disamenity;

- For incineration to AD: the figure which includes avoided external costs associated with electricity generated by both, but not incinerator disamenity.

Note that as with the changes in quantities, this does not assume a specific 'projection', so the results are not cumulative but indicative of the costs and benefits associated with the year 2010 situation. Table 50 gives the EU Totals at different growth rates.

Note that this assumes that all landfills are performing well. Were this switch to occur with landfills operating at their current gas collection efficiencies, the picture would look rather different (Tables 51 and 52).

In all cases, the situation shown is that in which waste diverted from landfill and incineration is diverted in a fixed proportion to compost and anaerobic digestion. This proportion reflects current practice and is set at a low figure of 5% for AD and 95% for composting.

The following observations can be made:

- The absolute magnitude of the benefits is affected by the growth rate of the stream. The higher the growth rate, the greater will be the benefits. This stands to reason. The more waste that needs to be treated, the greater the benefits of dealing with it in more benign and useful ways;

- Ideally, of course, growth rates are small. It is very important for this reason to understand that the growth rates may not be completely independent of the collection approach itself. Where source separation occurs, the potential for introducing instruments such as variable charging, designed to influence not only patterns of separation but also waste arisings, clearly increases. Hence, this – the effect in terms of public education / behavioural change - may be an important benefit of a requirement to source separate which is not being picked up here;[27]

[27] Equally, one can state that the way in which separate collection is enacted has important consequences for the total quantity of waste collected.

Table 49. External Benefits of Separate Collection Resulting From a Requirement for Source Separation (‘000 € in year 2010, well-behaved landfills)

		AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
From switch from landfill																							
	-1%	0	0	0	0	0	0	0	0	-4,522	4,413	0	0	746	3,148	154	891	111	4,363	0	0	0	0
	0%	0	0	0	0	0	0	0	0	-4,557	4,447	0	0	841	3,551	174	1,005	125	4,923	0	0	0	0
	1%	0	0	0	0	0	0	0	0	-4,595	4,484	0	0	903	3,814	185	1,070	141	5,547	0	0	0	0
	2%	0	0	0	0	0	0	0	0	-4,638	4,526	0	0	909	3,836	186	1,076	159	6,243	0	0	0	0
From switch from incineration																							
	-1%	414	571	832	1,133	1,516	2,084	2,117	3,304	4,144	15,022	5,322	9,887	0	0	0	0	0	0	70	79	1,866	2,327
	0%	468	644	939	1,278	1,710	2,351	2,388	3,728	6,267	22,718	6,004	11,155	0	0	0	0	0	0	79	89	2,106	2,626
	1%	527	726	1,058	1,440	1,927	2,649	2,691	4,201	8,637	31,308	6,765	12,569	479	934	173	340	0	0	89	101	2,373	2,959
	2%	593	817	1,191	1,621	2,169	2,982	3,029	4,728	11,280	40,886	7,614	14,147	1,702	3,319	549	1,078	0	0	101	113	2,670	3,330
TOTALS																							
	-1%	414	571	832	1,133	1,516	2,084	2,117	3,304	-378	19,435	5,322	9,887	746	3,148	154	891	111	4,363	70	79	1,866	2,327
	0%	468	644	939	1,278	1,710	2,351	2,388	3,728	1,711	27,164	6,004	11,155	841	3,551	174	1,005	125	4,923	79	89	2,106	2,626
	1%	527	726	1,058	1,440	1,927	2,649	2,691	4,201	4,042	35,792	6,765	12,569	1,382	4,748	359	1,410	141	5,547	89	101	2,373	2,959
	2%	593	817	1,191	1,621	2,169	2,982	3,029	4,728	6,641	45,413	7,614	14,147	2,610	7,155	736	2,154	159	6,243	101	113	2,670	3,330

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
From switch from landfill																					
	-1%	-42	-150	-1,364	-1,997	0	0	2,471	2,332	26	21	182	147	25	18	-407	-498	430	409	-32	-36
	0%	781	2,013	5,696	15,519	0	0	10,016	25,544	-79	77	377	1,332	63	266	587	3,388	65	2,554	146	293
	1%	832	2,144	5,730	15,609	0	0	10,311	26,295	-80	78	380	1,345	71	299	593	3,422	73	2,878	165	330
	2%	837	2,157	5,766	15,710	0	0	10,375	26,460	-80	78	385	1,360	80	337	597	3,444	82	3,239	186	371
From switch from incineration																					
	-1%	0	0	0	0	3,608	6,116	0	0	0	0	2,566	4,763	0	0	0	0	0	0	0	0
	0%	0	0	283	577	4,070	6,900	0	0	10	18	3,446	6,395	0	0	0	0	0	0	0	0
	1%	481	784	4,560	9,309	4,587	7,775	6,473	11,622	98	178	4,428	8,217	0	0	1,066	2,115	0	0	0	0
	2%	1,537	2,503	9,330	19,044	5,162	8,751	15,443	27,724	197	356	5,523	10,249	0	0	2,298	4,559	0	0	0	0
TOTALS																					
	-1%	-42	-150	-1,364	-1,997	3,608	6,116	2,471	2,332	26	21	2,748	4,910	25	18	-407	-498	430	409	-32	-36
	0%	781	2,013	5,979	16,097	4,070	6,900	10,016	25,544	-69	95	3,823	7,727	63	266	587	3,388	65	2,554	146	293
	1%	1,314	2,928	10,290	24,918	4,587	7,775	16,784	37,917	19	256	4,808	9,563	71	299	1,659	5,537	73	2,878	165	330
	2%	2,374	4,660	15,096	34,754	5,162	8,751	25,818	54,184	117	435	5,908	11,610	80	337	2,894	8,002	82	3,239	186	371

Table 50. External Benefits of Separate Collection Resulting From a Requirement for Source Separation ('000 € in year 2010) (switching ratios 95% to compost, 5% to AD, well behaved landfills)

EU	*Growth Rate*	Low	High
	-1%	20,233	58,347
	0%	42,006	123,390
	1%	61,123	164,542
	2%	85,229	215,046

Table 51. External Benefits of Separate Collection Resulting From a Requirement for Source Separation (‘000 € in year 2010, landfills with current gas collection efficiencies)

		AU		BE		DE		FI		FR		GE		GR		IR		IT		LUX		NL	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
From switch from landfill																							
	-1%	0	0	0	0	0	0	0	0	18203	30604	0	0	8860	12359	2523	3578	10581	16585	0	0	0	0
	0%	0	0	0	0	0	0	0	0	18342	30839	0	0	9995	13943	2847	4036	11938	18711	0	0	0	0
	1%	0	0	0	0	0	0	0	0	18497	31100	0	0	10734	14974	3031	4297	13452	21084	0	0	0	0
	2%	0	0	0	0	0	0	0	0	18671	31392	0	0	10799	15064	3049	4323	15140	23730	0	0	0	0
From switch from incineration																							
	-1%	414	571	832	1133	1516	2084	2117	3304	4144	15022	5322	9887	0	0	0	0	0	0	70	79	1866	2327
	0%	468	644	939	1278	1710	2351	2388	3728	6267	22718	6004	11155	0	0	0	0	0	0	79	89	2106	2626
	1%	527	726	1058	1440	1927	2649	2691	4201	8637	31308	6765	12569	479	934	173	340	0	0	89	101	2373	2959
	2%	593	817	1191	1621	2169	2982	3029	4728	11280	40886	7614	14147	1702	3319	549	1078	0	0	101	113	2670	3330
TOTALS																							
	-1%	414	571	832	1,133	1,516	2,084	2,117	3,304	22,347	45,627	5,322	9,887	8,860	12,359	2,523	3,578	10,581	16,585	70	79	1,866	2,327
	0%	468	644	939	1,278	1,710	2,351	2,388	3,728	24,609	53,556	6,004	11,155	9,995	13,943	2,847	4,036	11,938	18,711	79	89	2,106	2,626
	1%	527	726	1,058	1,440	1,927	2,649	2,691	4,201	27,135	62,408	6,765	12,569	11,213	15,909	3,204	4,637	13,452	21,084	89	101	2,373	2,959
	2%	593	817	1,191	1,621	2,169	2,982	3,029	4,728	29,950	72,278	7,614	14,147	12,500	18,383	3,598	5,401	15,140	23,730	101	113	2,670	3,330

		PO		SP		SW		UK		CYP		CZ		EST		HUN		POL		SLO	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
From switch from landfill																					
	-1%	5512	6355	27259	31310	0	0	17428	19682	630	723	4495	5155	652	746	8169	9459	10160	11707	656	762
	0%	7851	10413	27025	40133	0	0	26347	44212	319	537	1571	2723	747	1043	9596	13607	6193	9707	553	773
	1%	8363	11093	27182	40367	0	0	27121	45511	321	540	1587	2750	842	1175	9693	13744	6979	10938	623	872
	2%	8414	11160	27357	40627	0	0	27291	45796	323	544	1605	2780	948	1322	9754	13831	7855	12311	701	981
From switch from incineration																					
	-1%	0	0	0	0	3608	6116	0	0	0	0	2566	4763	0	0	0	0	0	0	0	0
	0%	0	0	283	577	4070	6900	0	0	10	18	3446	6395	0	0	0	0	0	0	0	0
	1%	481	784	4560	9309	4587	7775	6473	11622	98	178	4428	8217	0	0	1066	2115	0	0	0	0
	2%	1537	2503	9330	19044	5162	8751	15443	27724	197	356	5523	10249	0	0	2298	4559	0	0	0	0
TOTALS																					
	-1%	5,512	6,355	27,259	31,310	3,608	6,116	17,428	19,682	630	723	7,061	9,918	652	746	8,169	9,459	10,160	11,707	656	762
	0%	7,851	10,413	27,308	40,711	4,070	6,900	26,347	44,212	329	555	5,017	9,118	747	1,043	9,596	13,607	6,193	9,707	553	773
	1%	8,844	11,877	31,742	49,675	4,587	7,775	33,595	57,133	420	718	6,015	10,967	842	1,175	10,758	15,858	6,979	10,938	623	872
	2%	9,950	13,663	36,687	59,671	5,162	8,751	42,733	73,520	520	900	7,127	13,030	948	1,322	12,052	18,390	7,855	12,311	701	981

Table 52. External Benefits of Separate Collection Resulting From a Requirement for Source Separation (‘000 € in year 2010) (switching ratios 95% to compost, 5% to AD, landfills with current gas collection efficiencies)

EU	Growth Rate	Low	High
	-1%	137,585	194,313
	0%	151,095	249,156
	1%	174,839	295,671
	2%	202,291	350,068

- The total benefit being estimated is not enormous (of the order tens to hundreds of millions depending upon assumptions about growth rates, unit damage costs and the performance of landfills). This reflects:

    - The fact that many countries are already far advanced in respect of separate collection so the requirement to collect separately has relatively little effect when compared to their overall contribution to the waste stream (Germany, Austria, Belgium, Denmark, Netherlands); and

    - The fact that the net external costs of each unit of waste switched from one treatment to another is relatively low, especially where one assumes that most of the waste is diverted to composting rather than AD. Tables 53 and 54 show, for the purpose of sensitivity analysis, what happens when one considers a situation in which AD plays a more prominent role (for the cases where landfills behave well, and where their gas collection efficiencies operate at the current level). Here it is assumed that 40% of waste is diverted to AD and 60% to composting. The results are shown for the EU totals only. It can be seen that the benefits increase considerably. However, the private costs of this switch have to be considered also (see below).

Taken together, these points suggest that, notwithstanding the uncertainties surrounding the estimation of external costs of waste management, there may be some environmental justification for a policy supporting source separation. However, such a view should be contextualised by the costs of such a change.

Table 53. External Benefits of Separate Collection Resulting From a Requirement for Source Separation ('000 € in year 2010) (switching ratios 60% to compost, 40% to AD, well-behaved landfills)

EU	*Growth Rate*	Low	High
	-1%	38,703	121,491
	0%	58,545	176,407
	1%	79,965	225,241
	2%	106,698	284,505

Table 54. External Benefits of Separate Collection Resulting From a Requirement for Source Separation ('000 € in year 2010) (switching ratios 60% to compost, 40% to AD, landfills with current gas collection efficiencies)

EU	Growth Rate	Low	High
	-1%	156,055	257,458
	0%	167,633	302,173
	1%	193,681	356,370
	2%	223,761	419,527

7.2.3 Costs and Benefits of A Policy of Source Separation

The costs of implementing a policy of source separation in the Year 2010 are shown in Table 55. The key assumptions are as follows:

- Low and high costs of separate collection and composting of €35 / tonne and €75 / tonne;
- Low and high costs of separate collection and anaerobically digesting of €80 / tonne and €125 / tonne;
- Costs for landfill and incineration at €55 / tonne and €90 / tonne respectively. The landfill figure represents the estimated cost of landfill at a site meeting Landfill Directive requirements, and the estimated costs for incineration represent costs for a plant meeting Incineration Directive in a typical European situation where energy prices are not subsidised. This last point is important. Since the analysis already attributes external benefits to energy generation, revenues associated with energy production from waste imply double counting of the energy-related benefits;
- The switch from landfill and incineration is in the proportion 95% to composting and 5% to anaerobic digestion (which may be considered, in this analysis, to be the low cost, but lower external benefit, scenario); and
- The external benefits are represented for the case where landfills are well-behaved.

The numbers suggest that the costs of the policy change may be negative in low cost situations – substantially so (almost €1 billion) at higher growth rates – and approximately constant at €400 million in the high cost scenario. The former observation arises due to the avoided cost of higher cost treatments, these featuring more heavily as the growth rate increases (the proportion of the increase in separately collected waste which is being diverted from incineration increases as the growth rate increases).

At higher growth rates, the magnitude of the benefits approaches the magnitude of the costs as estimated in the ‘high cost’ scenario. In other words, for the high cost scenario, the benefit:cost ratio increases as the growth rate increases.

Table 55. Additional Costs, Avoided Costs, Total Private Costs and Costs Net of External Costs For A Source Separation Policy, €'000 in 2010, (switching ratios 95% to compost, 5% to AD)

Additional Private Costs of Biowaste Collection and Treatment		Additional Tonnage for Separate Collection	Additional Collection / Treatment Costs for Biowaste				
Growth Rate		(= A)	Low (= B)	High (= C)			
	-1%	23,352,935	869,897	1,809,852			
	0%	26,346,270	981,399	2,041,836			
	1%	29,687,637	1,105,864	2,300,792			
	2%	33,413,441	1,244,651	2,589,542			
Avoided Private Costs of Landfill and Incineration		Material Diverted from Landfill	Material Diverted from Incineration	Avoided Treatment Costs		Total Avoided Costs	
				Landfill @ (€55 / tonne)	Incineration @ (€90/tonne)		
Growth Rate		(=D)	(=E)	(= 55 x D/1000 = F)	(= 90 x E /1000 = G)	= (D x F) + (E x G) = H	
	-1%	20,102,907	3,250,029	1,105,660	292,503	1,398,162	
	0%	22,097,661	4,248,609	1,215,371	382,375	1,597,746	
	1%	22,965,402	6,722,235	1,263,097	605,001	1,868,098	
	2%	23,606,089	9,807,352	1,298,335	882,662	2,180,997	
Summary Cost-Benefit Analysis		Net Costs of Collection / Treatment		Total External Benefits		Total Costs Net of External Benefits	
Growth Rate		Low (= (B – H) = I)	High (=(C – H) = J)	Low (= K)	High (= L)	Low (= I – K)	High (= J – L)
	-1%	-528,266	411,690	20,233	58,347	-548,499	353,343
	0%	-616,348	444,090	42,006	123,390	-658,354	320,699
	1%	-762,234	432,694	61,123	164,542	-823,356	268,152
	2%	-936,346	408,545	85,229	215,046	-1,021,575	193,499

Even so, at high costs, the costs potentially exceed the magnitude of the benefits. Indeed, the range in estimated costs also exceeds the range in estimated benefits quite significantly. This is surprising at first sight. It suggests that the private costs of the change are not necessarily known with any greater certainty than the external costs. This may well be true given the range of parameters which may affect these costs, although it has to be re-stated that the external costs as used here do not cover the full range of estimates, and nor do any of the estimates cover the full range of external costs.

To illustrate sensitivity, two further situations are shown. The first is the situation were the prevalence of AD increases in the same way as was shown for the external costs (Table 56). This shows once again that the benefits are greater but it shows that these enhanced benefits come at a much increased cost.

The second is the situation where the private costs of incineration are increased to €100 per tonne and the private costs of landfill are increased to the point where the total costs net of external benefits are always positive even in the high cost scenario (Table 57). This situation arises when the landfill costs reach €71 per tonne.

This last situation represents an interesting scenario. It is the situation in which the policy effectively has a negative cost even in the worst case scenario. The point deserves to be made that landfill and incineration costs are not so different from these levels in those countries which are already furthest down this route. The suggestion is, therefore, that an alternative to an instrument requiring all countries to source separate waste materials would be a strategy in which the costs of residual waste treatments are raised to levels that make source separation the most desirable option from the financial point of view.

Such a policy has, effectively, already begun in the design of Directives on Landfill and Incineration. However, Member States may implement such legislation in different ways. To the extent that what is proposed is seen as desirable, the intention ought to be to ensure that Member States do not seek to implement these Directives in ways which are likely to reduce the costs of these treatments through side-stepping the intent of the Directives.

The effect of excluding the benefits of energy recovery in this analysis is shown in Table 58. The effect is not dramatic in absolute terms, although the benefits are doubled at lower growth rates. This is an important point since this change in benefits does not significantly affect the cost-benefit analysis. However, supporting the price of electricity by €0.02 per kWh electricity can generate considerable additional revenue for incinerators (of the order €10 per tonne of mixed municipal waste). To the extent that these revenues reduce costs by the same margin, they do have a significant impact on the overall analysis.

Table 56. Additional Costs, Avoided Costs, Total Private Costs and Costs Net of External Costs For A Source Separation Policy, €'000 in 2010, (switching ratios 60% to compost, 40% to AD)

Additional Private Costs of Biowaste Collection and Treatment		Additional Tonnage for Separate Collection	Additional Collection / Treatment Costs for Biowaste				
Growth Rate		(= A)	Low (= B)	High (= C)			
	-1%	23,352,935	1,237,706	2,218,529			
	0%	26,346,270	1,396,352	2,502,896			
	1%	29,687,637	1,573,445	2,820,325			
	2%	33,413,441	1,770,912	3,174,277			
Avoided Private Costs of Landfill and Incineration		Material Diverted from Landfill	Material Diverted from Incineration	Avoided Treatment Costs (€'000)		Total Avoided Costs	
				Landfill @ €55 / tonne	Incineration @ 90€/tonne		
Growth Rate		(=D)	(=E)	(= 55 x D/1000 = F)	(= 90 x E /1000 = G)	(= F + G = H)	
	-1%	20,102,907	3,250,029	1,105,660	292,503	1,398,162	
	0%	22,097,661	4,248,609	1,215,371	382,375	1,597,746	
	1%	22,965,402	6,722,235	1,263,097	605,001	1,868,098	
	2%	23,606,089	9,807,352	1,298,335	882,662	2,180,997	
Summary Cost-Benefit Analysis		Net Costs of Collection / Treatment		Total External Benefits		Total Costs Net of External Benefits	
Growth Rate		Low (= (B – H) = I)	High (=(C – H) = J)	Low (= K)	High (= L)	Low (= I – K)	High (= J – L)
	-1%	-160,457	820,366	38,703	121,491	-199,160	698,875
	0%	-201,394	905,149	58,545	176,407	-259,938	728,743
	1%	-294,653	952,227	79,965	225,241	-374,618	726,986
	2%	-410,084	993,280	106,698	284,505	-516,783	708,775

Table 57. Additional Costs, Avoided Costs, Total Private Costs and Costs Net of External Costs For A Source Separation Policy, €'000 in 2010, (switching ratios 95% to compost, 5% to AD) – 'Win-win' Scenario

Additional Private Costs of Biowaste Collection and Treatment		Additional Tonnage for Separate Collection	Additional Collection / Treatment Costs for Biowaste				
Growth Rate		(= A)	Low (= B)	High (= C)			
	-1%	23,352,935	869,897	1,809,852			
	0%	26,346,270	981,399	2,041,836			
	1%	29,687,637	1,105,864	2,300,792			
	2%	33,413,441	1,244,651	2,589,542			
Avoided Private Costs of Landfill and Incineration		Material Diverted from Landfill	Material Diverted from Incineration	Avoided Treatment Costs		Total Avoided Costs	
				Landfill @ (€ 71/ tonne)	Incineration @ (€100/tonne)		
Growth Rate		(=D)	(=E)	(= 71 x D/1000 = F)	(= 100 x E /1000 = G)	= (D x F) + (E x G) = H	
	-1%	20,102,907	3,250,029	1,427,306	325,003	1,752,309	
	0%	22,097,661	4,248,609	1,568,934	424,861	1,993,795	
	1%	22,965,402	6,722,235	1,630,544	672,223	2,302,767	
	2%	23,606,089	9,807,352	1,676,032	980,735	2,656,768	
Summary Cost-Benefit Analysis		Net Costs of Collection / Treatment		Total External Benefits		Total Costs Net of External Benefits	
Growth Rate		Low (= (B – H) = I)	High (=(C – H) = J)	Low (= K)	High (= L)	Low (= I – K)	High (= J – L)
	-1%	-882,412	57,543	20,233	58,347	-902,646	-804
	0%	-1,012,396	48,041	42,006	123,390	-1,054,403	-75,349
	1%	-1,196,903	-1,975	61,123	164,542	-1,258,025	-166,517
	2%	-1,412,117	-67,226	85,229	215,046	-1,497,346	-282,272

Table 58. External Benefits of Switch to Source-separation Assuming No Displaced Burdens ('000 € in 2010) (95% to composting, 5% to digestion)

EU		Low	High
	-1%	55,697	114,537
	0%	69,736	141,969
	1%	102,459	205,381
	2%	142,906	283,373

7.3 Effects of Implementing a System of Standards for Quality Composting

The impact of a system of standards clearly depends upon the scope and detail of that system. Such a system has the potential to impact on both private costs of composting and on the environment, although the higher end figures for costs are unlikely to be exceeded. It is important to recognise that some such standards may have impacts upon the environment that may not have been quantified in this work. For example, the imposition of heavy metal limit values will help to protect soil quality, but the incremental effect of such a change is beyond the scope of this modelling to detect.

The aims of systems of standards are, essentially, to combine the goals of:

- Giving confidence to end-users that the product they are purchasing meets certain quality standards designed to ensure 'fitness for purpose'; and
- Protection of soil, the environment and human, plant and animal health.

The two goals are largely complementary. It would be undesirable at this stage for the Commission to seek to establish, at the European level, statutory standards which address agronomic characteristics of composts. Individual countries are likely to have to establish fitness for purpose recommendations in conjunction with end-users. This should occur within the framework of standards which are intended to safeguard human health and the environment.

It would be difficult to understand the impact that this might have in terms of treatments without reference to the detail of such standards. However, three points seem important:

- The development of outlets for quality composts for utilisation in agriculture and horticulture is likely to be compromised in the absence of a system of standards. The key point here is that end-users may be dissuaded from using composts for the simple reason that they cannot be confident that the quality of the product they are using meets their needs;
- Standards which apply to processes can give confidence in terms of the elimination of certain organisms with potential to cause harm to humans, animals and plants; and

- Drawing distinctions between materials which are 'products' and materials that are 'wastes' makes clear that where certain qualities are not met, the resulting materials must be treated in line with legislation and controls which apply to waste materials. This means that unless such materials qualify for exemptions from permitting under Member State implementation of the Waste Framework Directive, they must be subject to licensing controls.

One can speculate as to the effects that a desirable system of standards might have on the different country groupings listed above. Group 1 countries would be affected only to the extent that existing systems of standards differed measurably from those proposed at the European level. Given the apparent success of these existing standards in providing a baseline standard for quality compost, significant departures are probably unlikely (although, equally, there is not universal agreement as to exactly what the complete system of standards should look like).

A system of standards should be helpful for Group 2 countries in giving confidence to end-users in what would be developing markets for quality compost. The design of source separation systems could be influenced by quality standards, recognising that the lower the rates of contamination, the more likely it becomes that the highest quality standards are met.

The same should apply to Group 3 countries, though to the extent that some of these might have anticipated making use of low-grade materials from mixed waste 'composting' as soil amelioration products, the re-definition of these materials as 'wastes' might discourage such treatments. Whether this would increase the degree to which source separation was implemented, or whether instead, the resort to incineration and other treatments increased as a result, cannot be determined in this analysis. Member State (and municipality) attitudes are clearly important here.

7.3.1 Quantification of Impacts of Implementing Standards Systems

It is difficult to quantify any impact resulting from this scenario. On the external costs side, the effects of changing the quality of composts cannot be quantified. This does not mean the effects are unimportant. Certainly, in Southern Member States, given the degree to which soil organic matter is, increasingly, a limiting factor in agricultural production, the temptation might be to make use of all organic material of whatever quality in an attempt to improve soil organic matter status. The risks associated with this would be that in solving one problem, one creates another (or others) related to heavy metals and, potentially, organic pollutants.

Standards can establish confidence in the minds of users of composts and those charged with dealing with the consequences of their application. To the extent that the lack of standards makes it difficult to develop the market for compost usage, standards have to be seen as integral to the process of ensuring that the benefits from compost utilisation can be captured.

It would be possible for a system of standards to have a very similar effect as a requirement for source separation. This would be the case in the following circumstances:

- There was a target for municipalities / provinces / regions for recycling and composting; and

- The standard effectively defined ‘compost’ in such a way that the quality required could only be achieved through source separation.

In these circumstances, the targets would effectively require source separation.

Such a standard could also have a ‘protective’ role in the sense that it would require that wastes which fell outside the standard still had to be treated as wastes. This would support a proper treatment of low quality outputs from biological treatment processes.

7.4 The Potential Impact of Agri-environmental Policies

Clearly the development of markets for compost is an important aspect of a system of standards. Furthermore, market development is necessary where source separation is made mandatory. EU experience thus far suggests that requirements for source separation need to be supported by standards for composting.

Further support for the marketing of compost could be given in those situations where it was felt appropriate to support the use of compost by farmers. The rationale for this might be different in different areas of Europe. Two important considerations in this respect might be the time-limited sequestration of carbon in soils, the implications of which are felt at a global level, and the related objective of building up soil organic matter content where this is already falling to low levels.

In Italy, support for the latter already occurs in three regions of the country. Payments made to farmers in Emilia Romagna amount to €155 per hectare for applications which build up soil organic matter on depleted soils. The Piemonte region pays €220 per hectare for applications of 25 tonnes dry matter per hectare applied over a five year time frame. The justification for such payments comes from Regulation EC 1257 on Sustainable Agriculture.

Discussions have already taken place in various international for a concerning the potential for awarding credits to farmers for the role played by composting in sequestering carbon in the soil. Similarly, in Canada, TransAlta has been seeking to acquire carbon credits through promotion of no-till soil management techniques which, because of the reduced disturbance experienced by the soil, do not lead to the same levels of mineralization of carbon as conventional agricultural techniques.

Regarding organic farming, the potential for enhanced use f composts seems, at one level, considerable. At the same time, however, organic standards bodies have displayed a great deal of concern regarding the potential for composts based on source-separated municipal waste feedstocks, to b contaminated by genetically manipulated organisms (GMOs). Until this issue is resolved, the use of composts from source separated municipal wastes is likely to be restricted to those derived from garden wastes.

7.5 Internal Market and Trade Issues

The study requires us to comment upon any significant potential internal market and trade issues that may arise as a result of different National legislation on composting, and to assess the impact of harmonised legislation on these issues.

Probably the most important internal market issue that could arise in the context of differences in National legislation is that which could occur through different Member States drawing different distinctions between 'wastes' and 'products'. This is a distinction which has statutory force in some countries. The movement of such materials across borders could clearly face some problems where one country's definition classified a material as product whilst the receiving country classified it as a waste.

Harmonising legislation in this regard would have clear advantages. This would enable compost products to marketed freely throughout the EU unencumbered by differing definitions as to when a 'waste' ceases to be a waste.

Beyond this issue, no new issues would seem to arise in the context of differing legislation on composting.

8.0 OBSERVATIONS, CONCLUSIONS AND RECOMMENDATIONS

8.1 Introduction

This report has sought understand, from a cost-benefit perspective, the implications of implementing a requirement for source separation of biodegradable municipal waste. The data requirements, and the information and technical demands placed upon those seeking to carry out such a study are immense. As those with experience in the field of waste management will know, the situation in respect of data is not good.

8.2 Key Observations

The following observations can be made:

1. Many countries already have quite well-developed systems for the collection of source separated biodegradable municipal wastes. These countries typically:

    a. Have relatively comprehensive systems of standards in place for compost. These standards usually seek to ensure protection of human health and the environment. They are supported by quality assurance systems for compost procducers;

    b. Treat the vast majority of this material through composting processes;

    c. Appear to have a strong motivation for doing so because the costs of residual treatments in these countries typically exceed the costs of separate collection-plus-composting;

2. For the EU as a whole, the effect of policy which seeks only to establish a Europe-wide system of standards, whilst of some utility, loses much of its transformative power unless:

    a. Member States have established targets for recycling and composting; and

    b. The targets, or the standards, or both draw a clear distinction between compost, defined as a product, and 'other materials', defined as wastes.

    Only in this situation would it seem likely that a 'standards only' policy would have a clear and positive effect;

3. Generally, the analysis of external costs and benefits is favourable to the separate collection and treatment of biowastes through composting or anaerobic digestion.

4. However, the magnitude of these benefits relative to other treatments is not large. This reflects the limitations of the economic analysis of external costs and benefits, which in turn. These, in turn, reflect the many limitations of the scientific knowledge concerning the actual effects one is seeking to quantify;

5. There is good reason to believe that a complete analysis of external costs and benefits of all waste treatment options would show compost and anaerobic digestion in a rather more positive light than has been the case in this analysis. This reflects the fact that some of the more negative aspects of landfilling and incineration have not been captured. Whilst the same can be said for some aspects of composting, these emissions tend to be relatively benign. Those that are potentially harmful are believed to be localised, and likely to affect only certain target groups. On the benefits side those associated with energy generation from incineration, landfill and anaerobic digestion are rather well understood. On the other hand, the benefits of compost utilisation are relatively poorly understood and not so easily quantified;

6. The private costs of different waste treatment options are not as well-established as one would like in carrying out this type of analysis;

7. In many countries (e.g. Austria, Germany and the Netherlands) the private costs of separate collection and composting are less than those of collecting residual waste and either landfilling or incinerating it. In other countries, the opposite may be the case (depending upon assumptions).

8. Whether the quantifiable benefits of the switch to separate collection and composting justify the costs depends upon how well-behaved alternative treatments are (do incinerators meet standards laid down in the latest Incineration Directive? Do landfills collect gas efficiently, and if so, do they generate energy from gas collected?). The worse these treatments behave, the greater the quantifiable external benefits of the switch to separate collection and biological treatment. Equally, the quantifiable benefits of the switch are likely to fall only once costs increase;

9. A number of Member State-specific policies affect the unit cost of the waste treatments being examined. These include policies on renewable energy (which can support prices for energy derived from waste), policies on packaging recovery and environmental taxes on landfill, incineration, transport and air emissions. In addition, Member States implement EU Directives, such as the Landfill Directive, the Incineration Directive and the IPPC Directive, in quite different ways. Because an increasing number of regulatory and economic instruments impinge upon costs, it is increasingly difficult to be clear about whether or not one is double counting, or benefits, in assessing the total private and external costs and benefits of the different approaches;

8.3 Conclusions

An important conclusion to be drawn from this study is that neither the private costs, nor the external costs and benefits, can be specified with much certainty. This implies that one has to treat the results with some considerable caution.

Regarding external costs and benefits, the analysis which has been carried out is, for all treatments:

- Incomplete (in the sense that it has not been possible to quantify all external costs and benefits);

- Affected by significant uncertainty; and

- Influenced by certain key assumptions which have material affects on results. It is unlikely that agreement could easily be reached on the correct nature of the assumptions.

Regarding private costs:

- The range of available technologies is increasing and each has different cost implications;

- Costs are affected by scale of treatment technology; and

- The structure of costs varies across countries, and within countries, owing to differences in policy at the Member State level (taxes, subsidies). Indeed, the costs vary over time as policy continues to evolve.

It seems reasonable to state that cost-benefit analysis alone cannot be used as a basis upon which to make decisions regarding waste management policy. The incomplete nature of such an analysis calls for other tools and criteria to support proposed changes in policy.

On the balance of evidence that has been presented, it seems that a policy of source separation will be justified where the collection system for source-separated biowastes is carried out in such a way as to optimise costs. Furthermore, where the costs of composting itself are kept to a reasonable level, it becomes likely that the net cost increase will be minimal, and may become negative (as is already the case in several countries) as costs for other treatments increase. It is worth noting that the costs of landfilling and incineration have shown a tendency to rise (owing to controls on pollutants etc.,) whereas those for enclosed composting and anaerobic digestion have, if anything, shown a tendency to fall. The costs for composting are likely to be lower under mandatory separate collection to the extent that this increases typical plant scale.

Other factors which may weigh in favour of this type of policy are:

- The fact that separate collection increases the possibility for implementing variable charging schemes, which can influence waste generation, and act to sensitise citizens to waste as an environmental issue;

- The relative unpopularity of larger treatments such as landfills and incinerators; and

- The potential linkages with agri-environmental / rural development policies, in which compost utilisation (and production) could be encouraged.

It is worth noting that if the private costs of the change to source separation were negative, this is a change which would be likely to occur anyway. It may well be the case that a 'policy' which seeks to reduce the level of the subsidies to incineration, and which seeks to ensure that the spirit and letter of the Landfill Directive is correctly applied (including, for example, issues related to the destiny of incinerator ash residues, and what should be seen to constitute pre-treatment) has a similar effect in terms of encouraging source separation as a mandatory requirement to implement this service. The separate collection of biowastes would flow naturally as an outcome of the relative costs of different waste management options.

The more difficult it becomes to ensure the relative costs of treatment options favour the source separation approach (for example, due to Member State initiatives to support the development of energy from waste as a renewable energy source), the stronger the argument becomes for implementing a *requirement* for source separation. This would constitute recognition of the fact that, however desirable (or not) the objective of supporting energy from waste might be in the context of a target for electricity generation from renewable energy, such policies have the potential to distort waste management decisions. This perspective is especially important given the 'so-far-limited' exploration of the potential external benefits of applying compost to the soil. In either context, the establishment of Europe-wide standards might enable compost products to be marketed more freely across Europe.

To conclude, a policy requiring source separation might not be necessary in a world where the Landfill Directive is fully implemented and where subsidies distorting the net costs of residual waste treatments were less prevalent than they are. However small the external benefits of such a proposed policy might appear (owing partly to the level of source separation already being achieved, but also, to the relatively low unit benefits of such a switch), these are benefits that can be captured at low or negative cost relative to the costs of alternative treatments. Where the net costs are not negative, they may be in the future.

It is quite possible, even likely, that the external benefits of applying compost to land will appear greater as understanding improves concerning the complex interactions between compost and soil. It should be borne in mind that the externality assessment leaves certain negative consequences of waste treatments under-explored, whilst the benefits side is relatively well-established for energy recovery facilities (being linked to the well-researched area of energy-related externalities). For compost, estimating the benefits has something more of an art about it. As such, over time, society may come to understand that the benefits of applying quality composts to the soil, and hence, of a policy mandating source separation, are much greater than have been anticipated thus far.

8.4 Recommendations for Future Investigations

There are several recommendations which flow from the above considerations:

a. A systematic attempt to understand the different influences on the costs of waste management options in the EU is urgently needed. For example, whilst the external cost assessment suggests that anaerobic digestion is a superior option in environmental terms, the assumption is that the higher private costs may not justify the additional external benefits. This conclusion would alter, however, if the costs of anaerobic digestion were lower than assumed here. Such a study should focus on the different economic and regulatory instruments which currently affect the relative costs of different waste management options in the EU. Regarding the Landfill Directive, this would require some check on the adequacy of financial provisions, an assessment of the mechanism through which it was ensured that full costs were passed on to waste producers (with some agreement as to the basis for ensuring this is done), and a review of the application of the requirements for pre-treatment of landfilled waste. These would affect the costs of landfilling. In addition, the way in which the end to co-disposal affects landfilling of incinerator ash residues will have an impact on the costs of incineration. Other instruments requiring investigation are the Incineration Directive, the IPPC Directive, Member State-specific taxes on landfill and incineration, and policies in place to support energy production from waste. The presence or absence of specific measures raises questions as to the degree to which an analysis of external costs and benefits, as carried out here, actually involves double counting of costs and benefits (since these may already be internalised in private costs).

b. Although this work constitutes the first attempt to quantify, in a comprehensive manner, the external benefits associated with use of compost, these estimates are subject to great uncertainty. They do, however, provide a reference point for further work which should seek, for example, to:

    i. Investigate further the potential for compost applications to contribute to the sequestration of carbon, and also, the build up of organic matter in the soil. Note that in conventional life-cycle approaches, which effectively ignore all biogenic carbon emissions, the scope for such an analysis is not possible owing to the assumptions made;

ii. Investigate further the potential of compost applications to reduce requirements for pesticide use in agriculture and other forms of cultivation;

iii. Understand more clearly the potential of nutrient applied to offset nitrous oxide emissions associated with the use of manures and synthetic nitrogenous fertilisers;

iv. Seek to quantify the external benefits associated with the reduction in the likelihood of flooding which might be occasioned by greater use of compost, owing to its tendency to support greater infiltration and retention of water;

c. The external cost analysis does not cover all impacts. Key omissions which could be addressed are:

i. For all treatments, the disamenity associated with the plant. This analysis has to be undertaken with great care, and indeed, whilst there is uncertainty concerning health effects of different treatments, an assessment of disamenity is unlikely to be straightforward, with hedonic pricing approaches potentially generating much lower estimates than contingent valuation methods;

ii. For all treatments, the impact of the treatments on operator health;

iii. For compost, as well as the positive aspects outlined above, on the negative side, the health effects of bioaerosols in the surrounding area, the impact of heavy metal applications to the soil (relative to alternative soil improvers), and the potential for impacts from any organic pollutants in compost; and

iv. For incineration, the impacts of all air pollutants as opposed to a sub-set thereof. The issue of non-chlorinated (e.g. brominated) dioxins and related compounds needs to be addressed. In addition, the impacts of various treatment routes for ash residues (including the use of residues in construction applications) need careful consideration.

v. For landfill, the external costs associated with leachate and the full range of gaseous emissions;