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FINAL REPORT

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

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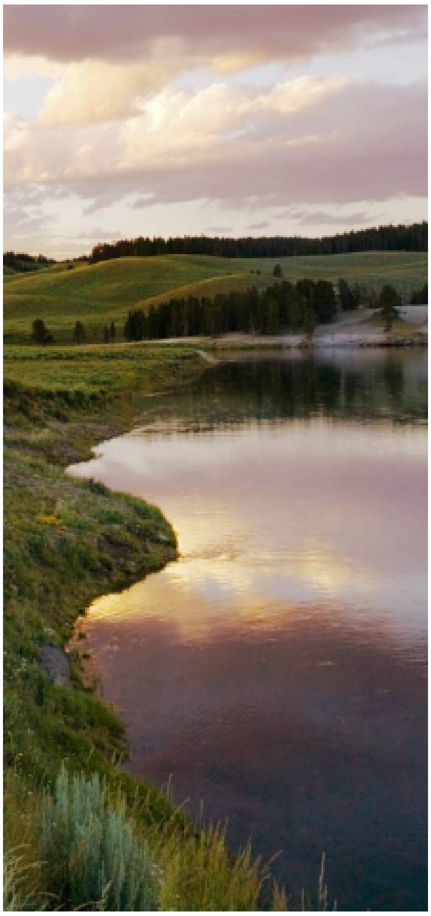

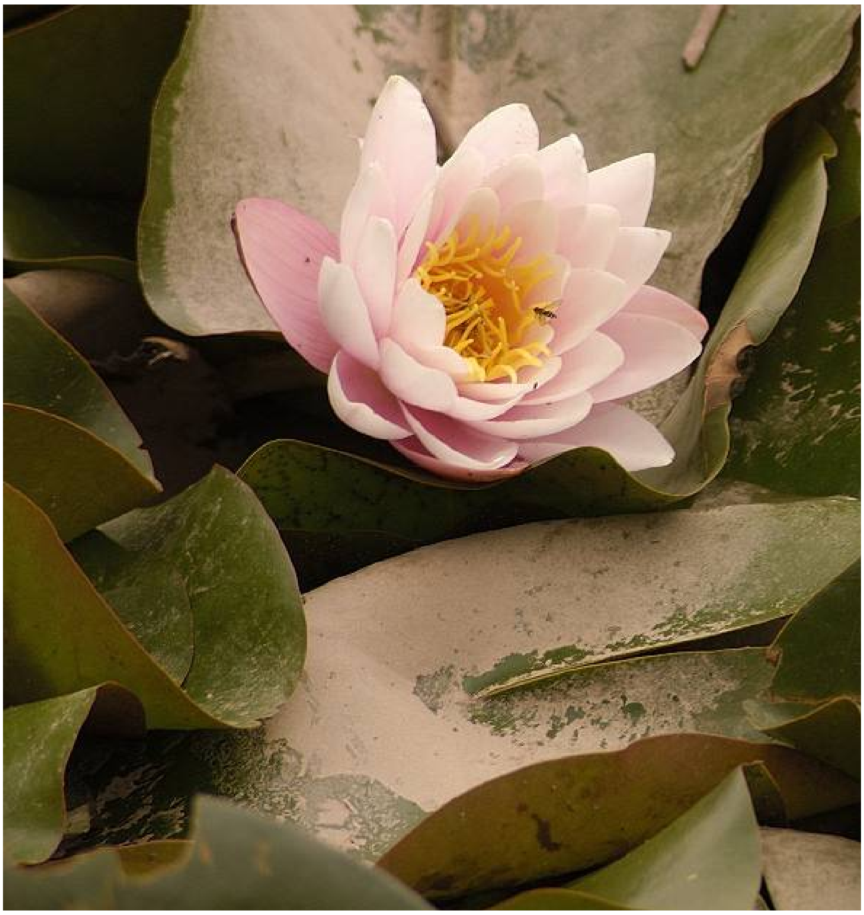

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ASSESSMENT OF THE OPTIONS TO IMPROVE THE MANAGEMENT OF BIO-WASTE IN THE EUROPEAN UNION

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LIST OF ABBREVIATIONS

AD	Anaerobic digestion
MBT	Mechanical biological treatment
ABPR	Animal By-Products Regulation
ACR+	Association of Cities and Regions for Recycling and Sustainable Resource management
BAT	Best available techniques
BIR	Bureau of International Recycling
BMW	Biodegradable municipal waste
BREF	BAT Reference Documents
CEWEP	Confederation of European Waste-to-Energy Plants
EEA	European Environment Agency
EEB	European Environmental Bureau
ETC/SCP	European Topic Centre on Sustainable Consumption and Production
FEAD	European Federation of Waste Management and Environmental Services
FGW	Fruit and garden waste
IPPC	Integrated Pollution Prevention and Control
ITT	Invitation to tender
LCA	Life cycle analysis
MSW	Municipal solid waste
VFG	Vegetable fruit and garden waste

EXECUTIVE SUMMARY

INTRODUCTION

The objective of this study was to look into ways of improving bio-waste management in the EU, and to provide an appropriate assessment of policy options, including the environmental, economic and social impacts, as well as prospective risks/opportunities. This work could lay the basis for possible additional measures at Community level in this area.

In particular, the project is expected to contribute to the Commission's assessment of the bio-waste management options. The final set of policy options to be assessed has been based on the results of the Commission's Green Paper consultations that were running in parallel to this study on the one hand and the preparatory work undertaken in the context of this study on the other hand.

The project work involved the following steps:

- Identification of relevant legislation;
- Data collection;
- Construction of a baseline scenario, involving projections of biowaste generation and treatment for all Member States until 2020;
- Identification of policy scenarios to be analysed;
- Estimation of the financial and environmental costs of each waste management option;
- Comparison of each policy scenario with the baseline scenario, in terms of financial costs and environmental impacts (with a special emphasis on green house gas emissions).

DELIMITATION OF THE SCOPE

The concept of bio-waste as used here is different from the concept of biodegradable waste as defined in the Landfill Directive. Indeed, the Waste Framework Directive defines bio-waste as “biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants”, whilst biodegradable waste is defined in the Landfill Directive (1999/31/EC) as "any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and green waste, and paper and paperboard". It was agreed with the Commission services that the emphasis of this project would be on kitchen and green waste streams.

The following technology options for biowaste management have not been considered in the study: pyrolysis, gasification, and food waste disposers.

RELEVANT EUROPEAN LEGISLATION AND POLICIES

Chapter 2 gives an overview of the most important European legislation and policies that affect the management of bio-waste. For the purposes of this study, the following issues have been especially important:

- The targets of the Landfill Directive regarding the reduction of biodegradable municipal waste (*not* biowaste) going to landfill;
- The emission limit values of the Waste Incineration Directive;
- The provisions on bio-waste and on end-of-waste criteria in the Waste Framework Directive;
- The Nitrate Directive, which imposes limits on N loads on farmlands, which can affect the application of compost to land;
- The EU policy for renewable energy, which affects the incentives for the use of bio-waste as a renewable energy resource.

Other relevant legislation (such as the Animal By-Products Regulation, the Packaging Directive, the proposal for a Directive on Industrial Emissions etc) is also discussed.

PREVIOUS WORK ON THE SUBJECT

Previous studies, such as those undertaken by Eunomia and COWI, have already taken the matter through the lens of cost-benefit analysis, and have explored the implications of different possible policy approaches.

The current study covers only bio-waste, which is a fraction of biodegradable waste (which also includes, for instance, paper and paperboard). This study also intended to cover industrial bio-waste (even though the inadequacy of the data have prevented use from conducting an in-depth analysis on this issue).

Another important change of scope is that the current study covers the EU27, rather than the EU15. This study has also to take into account several new legislative developments and uses the most recent estimates of the private and social costs of the different waste treatment options.

APPROACH TO DATA COLLECTION

The official reporting requirements with respect to the subject of this study are rather limited.

Member States have to submit the following information to the Commission:

- their waste management plan;
- their national strategies for the implementation of the reduction of biodegradable waste going to landfills.

Whenever these strategies have been put in the public domain, and have been made available in English or in French, they have been used as an input on this study. However, this situation was rather exceptional. Therefore, the project team has had to rely mainly on secondary sources.

Information from the European Topic Centre on Sustainable Consumption and Production ETC/SCP, Eurostat and the European Environment Agency (EEA) has been complemented with a variety of other data sources, mainly reports from national environment ministries, waste management agencies and statistical agencies.

For each country, we have drafted a “fact sheet”, containing all the information we have identified. In some countries, there is a very significant discrepancy between some of the figures provided by the ETC/SCP and data from national sources. However, even national sources are not always consistent.

For the construction of the baseline, a choice between conflicting figures had to be made, often based upon expert judgement from the project team. The figures that were eventually used for the construction of the baseline are always reported.

Chapter 4 of the main report discusses the most inconsistencies and uncertainties with respect to individual data sets in detail.

METHODOLOGY FOR THE BASELINE SCENARIO

As a first step in this assessment, we have constructed a baseline scenario with respect to the generation of bio-waste, its collection and its treatment for all 27 Member States. Trends are calculated until 2020.

As requested in the Terms of Reference to this study, the baseline scenario is developed on the assumption that all Member States are coping with the targets from the Landfill Directive and with the policy targets they have imposed on themselves. This will request for many Member States a considerable and persistent effort on developing alternatives for landfill, on top of what is already undertaken. The baseline scenario should therefore not be considered as a business-as-usual scenario, as there is a risk that business-as-usual may not lead to compliance with the Landfill Directive in some Member States.

It has been argued by some that a Biowaste Directive could be necessary as a tool to support Member States in reaching the targets. This is a political question that we do not consider.

The impact assessment that is developed as a comparison of the baseline scenario with other possible scenarios is an exercise to see what the costs and the benefits of the different scenarios will be, assuming that in the Baseline, the Landfill Directive targets are reached. As such, under anticipated Baseline scenarios for separate collection of biowaste, the Landfill Directive targets imply a given minimum quantity of treatment of biowaste as residual waste through means other than landfill. Whether these targets can be reached in the absence of a Biowaste Directive falls outside the scope of this study.

With the above, the impacts and potential benefits of a Biowaste Directive may be anticipated to exceed those as shown within this report

Generation and composition of municipal waste

The baseline scenario concerning the generation and composition of municipal waste is based upon:

- The actual generation of this waste fraction in 2006
- The demographic evolution
- The evolution in GDP, as an indicator for changing consumption patterns.

We have divided Member States into three different classes upon which different scenarios can be applied.

- A scenario 1 where in a first phase, due to quick economic growth and a catch up operation in a context with less environmental awareness or pressure, waste generation grows more quickly than the economy. This first phase is followed by stabilisation.

- A scenario 2 where no decoupling takes place and the environmental impact evolves at the same speed as economic activity.
- A scenario 3 where the waste generation is decoupled from the economic growth (relative decoupling) and tends to stabilise around a maximum value. The only factor influencing the waste quantity is the demographic growth.

For each Member States, an estimate has been made of actual coverage of the collection system.

Estimates have also been made of:

- Composition of mixed household waste and its content of bio-waste;
- Share of biowaste in total municipal biodegradable waste;
- The share of food waste and garden waste in biowaste generation;
- The share of food waste and garden waste that are collected separately.

As the existing harmonised reporting requirements at the EU level are not adapted to the data needs of this study, data on the subject at Member State level is sometimes sketchy, not always accurate, and definitions and reporting methods are not harmonised across MS. This means that a wide range of uncertainty surrounds the starting point of the baseline construction.

The following table summarizes the projection of MSW generation at the EU27 level:

	EU27		
	Biowaste	Non-biowaste	Total MSW
2008	87.718.367	162.509.636	250.228.003
2009	89.020.010	164.362.841	253.382.851
2010	90.026.473	165.913.194	255.939.667
2011	90.914.846	167.127.565	258.042.411
2012	91.801.186	168.322.808	260.123.994
2013	92.549.266	169.333.957	261.883.223
2014	93.219.592	170.241.734	263.461.326
2015	93.847.413	171.036.366	264.883.779
2016	94.418.616	171.756.911	266.175.527
2017	94.988.912	172.459.206	267.448.118
2018	95.558.307	173.159.548	268.717.855
2019	96.070.837	173.785.870	269.856.707
2020	96.582.909	174.410.947	270.993.856

Biowaste treatment

Next, projections have been made of current and future distribution of **bio-waste treatment** over:

- Landfilling;
- Incineration;
- Mechanical Biological Treatment (MBT) with aerobic or anaerobic biological treatment;
- Composting;
- Anaerobic direction;
- Home composting.

Specific treatment methods for bio-waste only apply to the fraction of separately collected waste. For the fraction of bio-waste included in the mixed household waste fraction, only landfilling, incineration or MBT can be considered.

The projections have been based upon the preferences of the Member States as expressed in their waste management plans if publicly available.

In the absence of publicly announced policies, we have used expert judgement to extrapolate from similar countries.

The following table summarizes the projection of MSW treatment methods at the EU27 level:

	Landfill	Incineration	MBT	Composting	Home composting	AD
2008	35.738.844	17.402.551	11.211.481	18.658.979	654.508	1.534.937
2009	35.171.916	17.728.458	11.473.431	19.642.019	724.430	1.722.447
2010	33.998.654	18.377.499	12.066.505	20.266.791	797.920	2.258.470
2011	32.240.606	18.619.624	13.697.095	20.949.277	883.454	2.767.027
2012	29.689.359	19.420.645	15.493.077	21.298.551	966.487	3.654.014
2013	24.346.536	20.513.166	19.652.181	22.123.592	1.046.651	4.073.340
2014	22.832.121	20.764.563	20.778.080	22.908.972	1.120.475	4.457.254
2015	21.635.520	21.401.242	20.936.875	23.674.209	1.197.558	4.857.487
2016	19.247.054	21.631.446	22.444.891	24.430.271	1.277.907	5.272.295
2017	18.440.031	22.029.764	22.198.214	25.168.652	1.355.195	5.703.790
2018	17.837.002	22.032.829	22.113.687	25.962.491	1.439.964	6.103.070
2019	16.651.462	22.584.068	21.989.103	26.748.914	1.530.648	6.520.225
2020	15.121.568	22.552.781	22.771.721	27.600.154	1.626.861	6.885.235

DEFINITION OF THE POLICY SCENARIOS

All policy scenarios that have been withheld for further analysis share a series of common assumptions:

- The compost that will be generated in this policy scenario will fulfil the same quality requirements as described in policy scenario 1 (see below)
- The amount of bio-waste *recycled* is assessed as the sum of
  a. The input of bio-waste for compost production minus the rejects and recycling residues
  b. The input to AD minus rejects and recycling residues, assuming that the digestate will be used as a recycled product or that it will be further composted and used as compost.
  c. The input of bio-waste for home composting
- Bio-waste can be considered as recycled under the condition that the compost or digestate generated at the end of the process will effectively be used or marketed as a product.
- Only waste that is collected source separated can be recycled.
- Use of digestate or compost that does not fulfil the quality standards for free use, and that is applied as an intermediary or final landfill cover, is not considered recycling.
- The output of the MBT process can no longer be considered as bio-waste.

- Home composting is a recycling method which does not require collection. It is considered recycling under the conditions that the home composting programme is well funded, well structured and monitored. Allowing home composting to be included in the recycling target accommodates the needs of areas with low population density.
- We assume that the quantities of food and garden waste that go to every recycling option never drop below the quantities assumed in the baselines, except due to waste prevention. In other words, it is assumed that no switch from AD to composting (or vice versa) takes place: all changes in the quantities are due to increases in selective collection.
- The following timeframe is set for the recycling targets and for prevention.
    a. No deviation from the baseline is expected as long as the recycling targets do not enter into force.
    b. As starting date for the deviation from the baseline policy scenarios, we will take 2013. 2017 would then be the date for the interim targets and 2020 the date for the final target.
    c. We assume that four years after the entry into force, an interim target has to be met which corresponds to 40% of the distance between the start value in 2013 and the final target.
    d. We assume that progress between the different targets will be piecemeal linear.

The specific assumptions of each policy scenario are described below.

COSTS AND BENEFITS OF THE APPROACHES TO BIOWASTE MANAGEMENT

We have modelled, for individual Member States, ‘tonne for tonne’ comparisons for the different approaches to biowaste management, highlighting the financial costs and the environmental costs and benefits, and the reasons for variation therein. We have also provided a rough estimate of direct employment effects of changes in waste management methods.

We have modelled financial costs under both a ‘private’ and a ‘social’ cost metric:

- The cost of capital used for the private cost metric ranges from 10% to 15%, depending on the process. The private cost metric also includes the relevant taxes, subsidies, support mechanisms that apply to the management of waste.
- We have used the European Commission standard discount rate for impact assessments for the social metric at 4%. Under the social metric, the discount rate is used as the cost of capital. The calculation of financial costs under the social metric does not include the effect of taxes and subsidies.

In order to adjust the capital and operational expenditure across the EU, we have allocated a percentage of labour cost related to the assumed operational and capital expenditure. The labour cost per country has been varied based on an index of labour rates.

The discussion covers:

- Food waste prevention;
- Home composting;
- Collection methods (including separate collection);
- Composting and AD (including electricity production, vehicle fuel substitution and gas to grid variants);

- Incineration (including electricity only and combined heat and power (CHP) variants);
- Landfill;
- Switching from landfill to organic treatment systems;
- Switching from incineration to AD; and
- Switching from incineration to AD where the capital costs are assumed to be zero (as a proxy for the case where costs in incineration are 'sunk').'

The main findings are:

- Home composting might not deliver the same environmental benefits as anaerobic digestion. However, in the overall analysis of costs and benefits, the financial cost savings are likely to outweigh the reduced benefits from not collecting waste for anaerobic digestion. Therefore, net welfare gains are to be made from home composting.
- The costs of particular services can be variable depending on methods of collection. However, we have shown that biowaste collection (in the context of integrated and optimised collection systems) can be undertaken with zero additional financial cost and without significant increase in transport related externalities. As such, we do not value the contribution from collection costs and impacts within the results in the modelling through the remainder of this report.
- The issue of how countries meet recycling targets depends very much on the level on that target and which collection systems are assumed to be in place in the future. Targets demanding a simple overall recycling rate of for instance 50% may be achievable by simple incremental adaptation of existing garden waste collection systems to incorporate food waste. Higher targets are more likely to require a move to independent collection systems.
- The environmental benefits of waste prevention (as opposed to home composting) are potentially very important. Moreover, it is likely to bring financial benefits as well, even if one takes into account the costs of waste prevention campaigns (especially in the case of food waste prevention).
- Whilst free garden waste collections can increase quantities collected, food waste collections (especially when they are accompanied by reductions in refuse collection frequency) may lead to a reduction in food waste generation in the first place. This is of particular significance given the previous point.
- The low financial cost of IVC, combined with its relatively low external environmental cost, make it the best performing option for 19 out of 27 Member States under the private metric. The relative environmental benefits associated with the AD options continue to be insufficient to outweigh the influence of cost in the majority of cases under this metric.
- For both IVC and AD, the current source of electricity is an important factor in determining the net benefits of different switches. The biogas to vehicle option typically performs the best of the different AD options in those countries that generate their energy from the cleaner sources of electricity.
- There is a net cost to society in closing down existing incineration capacity to build new biowaste treatment facilities if one ignores the capital costs of exiting incineration capacity (in other words, if one considers the existing investment in incineration capacity to be *completely* "sunk"). In reality, a number of additional factors would serve to reduce the extent to which the issue of sunk costs becomes problematic:

a. The problem of switching from incineration to other treatments is only a potential problem for capacity *that already exists*. Since the policies proposed take effect over an extended period of time, it would be expected that developers would adjust their decisions to invest in additional incineration capacity or to undertake upgrades /retrofits to facilities which are reaching the end of their useful life;

b. This would then mean, assuming a lifetime of 20 years for incinerators, that some existing incinerators could be phased out over the time during which the policy took effect, or otherwise, that replacement facilities could be sized accordingly. Therefore the above situation, in which the extreme case of a zero cost for capital was implied, would most likely be associated only with capacity already, or soon to be, in place, and then, the problem would be reduced for existing facilities owing to the time over which the policy would take effect.

- A change in waste management away from incineration to composting could lead to the *direct* creation of a few thousands jobs at the EU27 level in composting activities and maybe a few tens of thousands jobs in waste collection. However, the employment effects described above will, to some extent, be “crowded out” at the macro economic level. The only assumption under which the scenarios described above could lead to net job creation in the wider economy is if the people employed in waste management would not be competitive on the regular labour market.

POLICY SCENARIO 1

Compost standards represent a valuable quality framework within which other policy measures can more successfully be enacted. As such, it will be assumed that any additional option under analysis is combined with a compost standard. However, we have also assessed compost standards as “stand alone” instrument, in order to have a better understanding of the value added of the other instruments under consideration.

Composts derived from source segregated materials are likely to have much lower levels of contamination from potentially toxic elements, such as metals and organic pollutants. It is very difficult to place a monetary value upon the environmental benefits, other than through seeking to understand clean-up costs should this situation emerge. It might be noted, however, that prices paid for quality compost are typically zero (in agriculture) or positive, whereas prices are often negative for materials derived from mixed waste (the producers of these materials pay to ensure an outlet).

In addition, standards, when coupled with Quality Assurance Systems (QASs), can assist in the development of markets for compost.

As regards Scenarios 2 and 3, but especially with regard to Scenario 2, it seems unlikely that such a rapid development of source separation and composting could materialise without standards and QASs being developed in the countries concerned. The swift development of a standard might even be a pre-condition for the baselines to be achieved.

However, at the EU level, a ‘weak’ standard might be less valuable than none at all. Allowing products of low quality to be marketed without differentiating them from higher qualities can be confusing for consumers and risks jeopardising the development of a market for compost, and a positive image for it.

SCENARIO 2 “HIGH PREVENTION AND RECYCLING”

This scenario can be interpreted as the “high ambition scenario”, characterised by important waste prevention and high recycling rates.
This scenario includes:

- Waste generation will be reduced by 7.5% compared to the baseline, as a result of effective waste prevention.
- The costs of the preventive actions will not be included quantitatively in the model, as it is not defined which preventive actions will be taken at which cost in which country. Moreover, as note above, whilst there may be costs to central organisations in organising campaigns, the combination of savings accruing to private households, as well as avoided disposal costs, make it likely that the net financial costs will be negative);.
- Home composting will take place in the same proportion as in the baseline policy scenario, it will contribute to reaching the recycling targets.
- The targets for separate collection are 60% of kitchen waste (food waste) and 90% of green waste.
- It is assumed that all garden waste that is collected separately *in addition* to the baseline is recycled in IVC.
- It is assumed that all food waste that is collected separately *in addition* to the baseline is sent either to IVC, or to AD, depending on the recycling option that yields the highest net benefits to society. “Net” benefits include financial costs, but also environmental costs, including those related to the emissions of GHG.

Scenario 2a is identical to scenario 2, except that all food waste that is collected separately in addition to the baseline is sent to AD, which is the recycling technology that yields the highest benefits in terms of GHG emission reductions. Under scenario 2a, there is thus a much higher focus on the potential of biowaste recycling as a tool to reduce GHG emissions.

SCENARIO 3 "LOW RECYCLING"

Scenario 3 and 3a follow the same general approach as set out for scenario 2, with two differences:

- For the definition of the “low” recycling target for 2020, we have used the midpoint between the current EU27 average and the current rate in the member state with the highest recycling rate.
- Prevention will not be considered.

Thus, compared to scenarios 2 and 2a, this scenario is much less ambitious.

SUMMARY OF POLICY SCENARIOS 2 AND 3

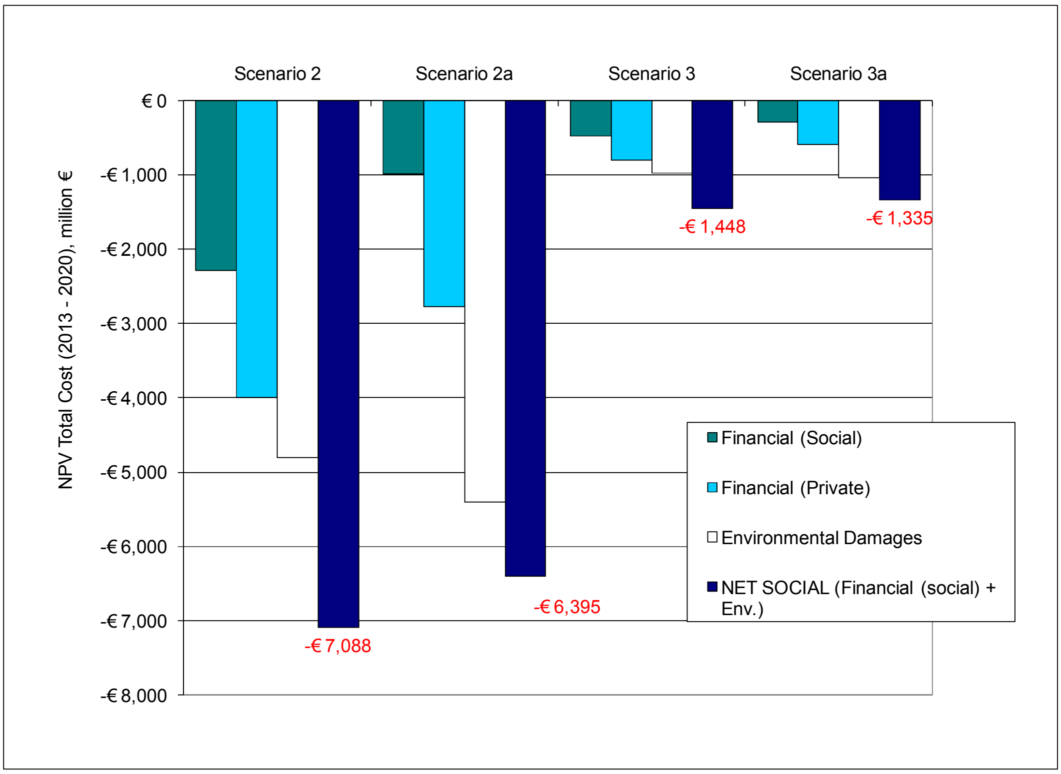
Net Cost to Society

The measure most often used in cost benefit analyses and impact assessments is the net cost to society. This is comprised of environmental damage costs and financial costs. In this instance the financial damage costs are those calculated with respect to the social cost metric. It is this figure which is used to determine whether the impact of a policy is positive, or negative, with respect to society.

To understand where the figure has come from, its' main component parts are included in the charts. The financial costs under the private metric are also included, for consideration, as it is these costs that will be seen in the market place. The net present value of both the resultant change in waste management from 2013 to 2020 and the final situation in 2020 is presented. This provides evidence as to the combined effect of the policy over the modelled period (i.e. 2013 to 2020), as well as the further annual benefit that could be achieved through the resultant increase of waste treatment infrastructure required to be in place by 2020, for the targets under each scenario to be met.

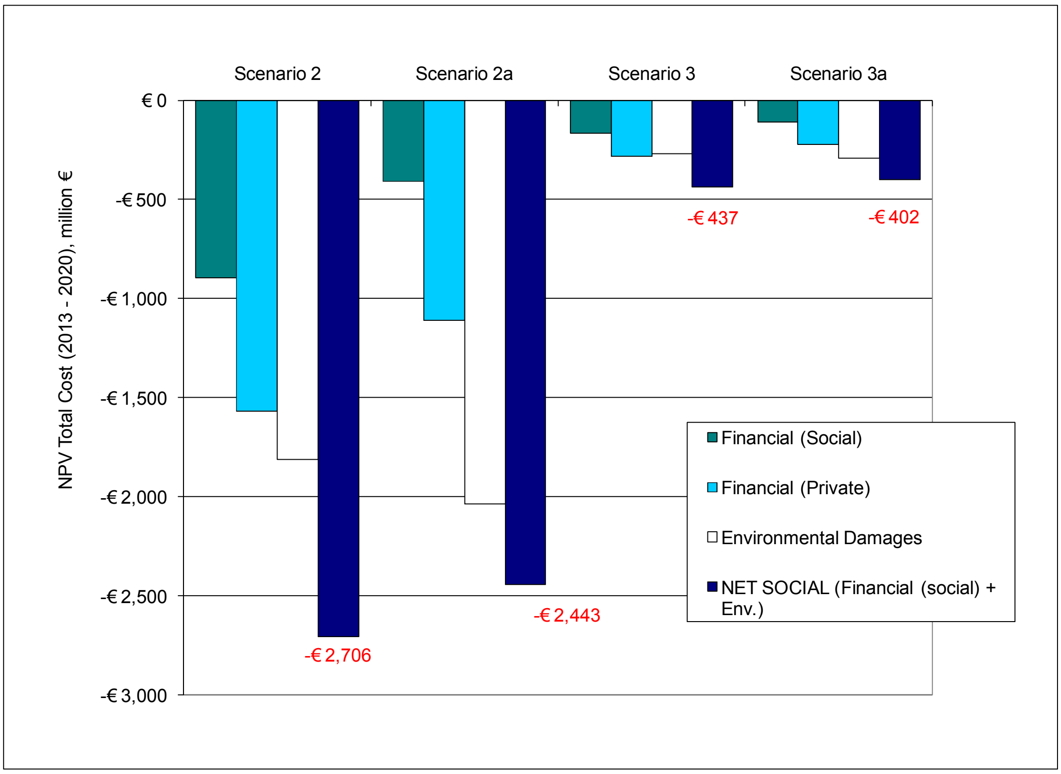
The figure below shows that under all scenarios there is a *significant* net benefit to society. The greatest benefit is under Scenario 2, at just over € 7 billion - in net present value terms. However, under Scenario 2a, where the best GHG performing biowaste treatment options are considered for each country, the environmental benefits are further increased by 12% (though financial costs are higher).

Financial and Environmental Costs of Each Scenario for the EU-27 – Total Cost/Benefit (NPV), million €



The figure below shows that the additional treatment of source separated biowaste develops a significant annual benefit to society from 2020 onwards. This is of great importance since, given that nearly 40% of the total benefit occurs in 2020, the continued benefits, beyond the period modelled in this study, will remain significant.

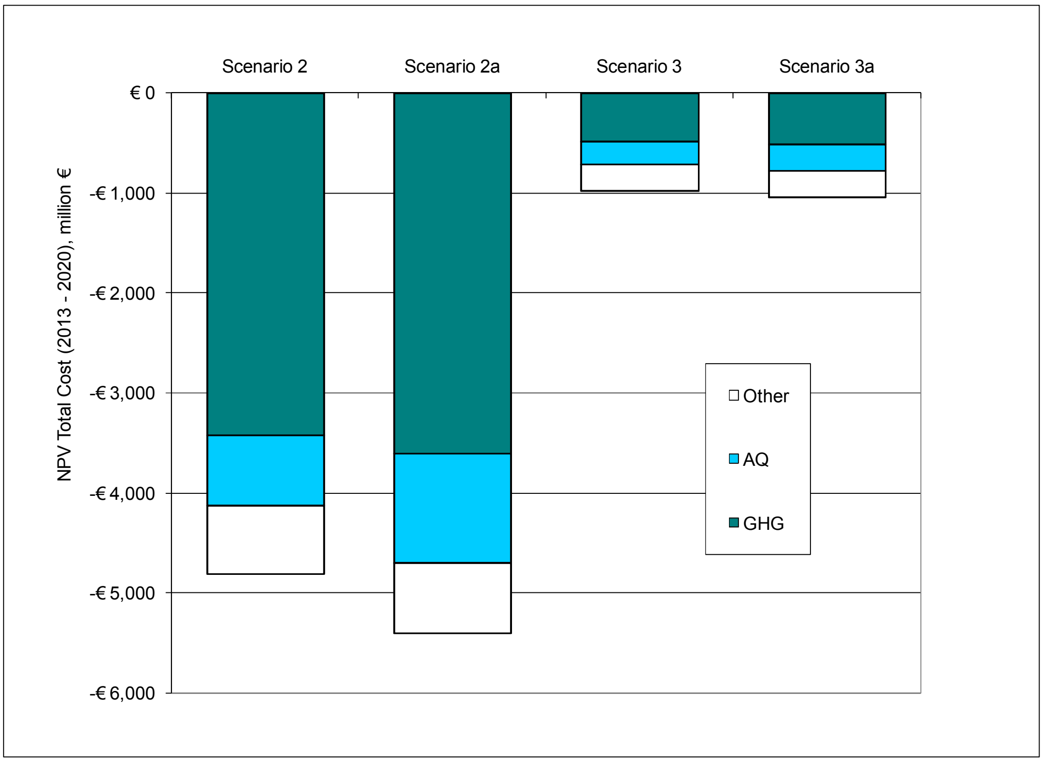
Financial and Environmental Costs of Each Scenario for the EU-27 – 2020 Only
Cost/Benefit (NPV), million €



Environmental Damage Costs

In order to understand what factors are the most significant in the determination of the net cost to society, it is important to be aware of the component environmental damage costs. The figure clearly shows that the greatest proportion of the benefit to society accrues from the reduction in greenhouse gas emissions. Given that the cost of carbon is likely to increase over time, not decrease, the current snapshot of estimated benefits to society are expected to be understated.

Environmental Damage Costs of Each Scenario for the EU-27 – Total Cost/Benefit (NPV), million €



Total Greenhouse Gas Implications

The commitments of the Community, as endorsed in the European Council of March 2007, are:

- to achieve at least a 20 % reduction of greenhouse gas emissions by 2020 compared to 1990;
- to achieve a 30 % reduction of greenhouse gas emissions by 2020 compared to 1990 provided that other developed countries commit themselves to comparable emission reductions and economically more advanced developing countries commit themselves to contributing adequately according to their responsibilities and capabilities.

With a 20% target, this means that emissions should be reduced to from 5 564 to 4 451 million tonnes CO_2 equivalents. With a 30% target, emissions should be reduced to 3 895 million tonnes CO_2 equivalents[1].

In the EEA report, two projections are used for 2020:

- In the "with existing measures" (WEM) projections, EU27 GHG emissions in 2020 will be 6% below 1990 levels (5 230 million tonnes CO_2 equivalents); this means that under WEM, the 20% target is exceeded by 779 million tonnes CO_2 equivalents;
- In the "with additional measures" (WAM) projections, EU27 GHG emissions in 2020 will be 14% below 1990 levels (4 785 million tonnes CO_2 equivalents) this means that under WAM, the 20% target is exceeded by 334 million tonnes CO_2 equivalents;

For the purposes of this study, we shall limit ourselves to the WEM projections.

[1] For the 1990 emissions, we have used Table 2.4 in: EEA, Technical report No 04/2009, *Annual European Community greenhouse gas inventory 1990–2007 and inventory report 2009; Submission to the UNFCCC Secretariat*; Version 27 May 2009.

The following table summarizes, for each policy scenario, the reductions of GHG emissions in 2020 compared to the baseline, expressed in million tonnes of CO_2 equivalents.

Table 1-1: reductions of GHG emissions in 2020 compared to the baseline

	Including biogenic CO_2	Excluding biogenic CO_2
Scenario 2	48.8	40.1
Of which: waste prevention	*32.6*	*32.6*
Change in biowaste management	*16.2*	*7.5*
Scenario 2a	51.5	44.2
Of which: waste prevention	*32.6*	*32.6*
Change in biowaste management	*18.9*	*11.6*
Scenario 3	4.3	1.9
Scenario 3a	4.8	2.6

In other words, if we exclude biogenic emissions, the reductions in GHG emissions under policy scenario 2a would correspond to 6% of the current difference between the 2020 WEM projections and the 2020 targets. About 74% of this contribution would be due to waste prevention effects only. Under policy scenario 2, the reduction in GHG emissions is only slightly smaller than under scenario 2a, mainly because the (large) waste prevention effect is the same under both variants.

In the case of policy scenarios 3 and 3a, the possible reduction in GHG emissions is more than an order of magnitude smaller (0.25% and 0.3% respectively).

As our results indicate that the financial costs of the policy changes are negative (in other words, that the analysed policy changes all induce financial costs savings), these emission reductions can be considered to be “low hanging fruits” in terms of climate policy. However, due to differences in calculations methods used in this study and UNFCCC reporting requirements, these estimates cannot be used directly to assess contribution of biowaste policy to meeting formal GHG reduction targets.

LIMITS TO THE ANALYSIS

In the interpretation of the results above, it is important to keep in mind that they are the result of modelling work, and that each model always is a simplification of reality. Although the limitations of our analysis have always clearly been explained in the text, we think it is important to repeat the most important ones:

- There are important **gaps in the data** reported by the Member States. Therefore, there is a lot of uncertainty surrounding even the current state of biowaste generation and treatment.

- There is a lot of **uncertainty concerning the policy intentions of some Member States.**
- The baseline scenario assumes **compliance with the Landfill Directive**. However, as acknowledged by the Commission (see COM(2009) 633 final) several Member States are not currently moving in the right direction.
- There is a lot of **uncertainty regarding the parameters that will influence future costs and benefits** (GDP growth, electricity prices, assumptions regarding financial costs, the choice of damage costs used to assess the pollutants, and other factors specific to different treatment methods)
- Every model involves a trade-off between detail and realism on the one hand and tractability and transparency on the other hand. This requires some aggregation at the geographical and technical level.
- The benefits we have reported have been derived under a central case which assumes the change in collection costs can be constrained to zero. There are good reasons why this assumption would be likely to hold good where collection systems are well designed. However, we note that where collection systems are poorly designed, then additional financial costs may be incurred, and these may eliminate the benefits estimated in the central case.

POLICIES

Our analysis has estimated the costs and benefits of reaching some uniform prevention, collection and recycling targets. It has not considered the policies that would be needed at the Member State level to implement these targets.

Possible policies that have already been used in other Member States could include:

- Ordinances for separate collection, requiring local authorities to organise separate collection;
- Targets for recycling and composting / digestion;
- Targets for reducing residual waste;
- Food waste prevention campaigns; and
- Landfill / incineration taxes and bans (which lend support to, rather than drive, such measures)

Subsidiarity implies that while the EU sets the framework, Member States would be free to implement the targets as they see fit in line with subsidiarity.

This could also imply a differentiation in the timeline for the targets. For instance, in order to accommodate the needs for incinerator heavy countries, any policy could incorporate a derogation period for such countries, much as happens with the Landfill Directive targets in cases where Member States have, historically, landfilled very large proportions of municipal waste.

1 Introduction

1.1 Background

One of the Commission objectives in the area of waste management is to improve the way in which bio-waste is managed in the EU.

The Environment Directorate-General (DG ENV) of the European Commission is therefore currently carrying out an assessment of the existing and possible future management options of bio-waste. This assessment includes:

- launching of a Green Paper “On the management of bio-waste in the European Union” (COM(2008) final of 3.12.2008), followed by a stakeholder consultation
- preparation of an impact assessment
- possible legislative or non-legislative follow-up.

The concept of bio-waste as used in this study is more restrictive than the concept of biodegradable waste as defined in the Landfill Directive. Indeed, the Waste Framework Directive defines bio-waste as “biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants”, while biodegradable waste is defined in the Landfill Directive (1999/31/EC) as "any waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and green waste, and paper and paperboard".

The Commission's assessment will take into account the following existing measures applicable to bio-waste, in light of the information and recommendations received from the stakeholder consultation and the current study contract:

- The proper waste management of bio-waste in the EU has been addressed as one of the priorities in the Thematic Strategy on the Prevention and Recycling of Waste (see Section 2.4 for more details).
- The issue of bio-waste management is already being, or will in future be, addressed in a number of Community actions, including the Landfill Directive (Section 2.1), the Regulation on Animal By-Products (Section 2.3), the revision of the IPPC Directive (Section 2.6), the ongoing work on the end-of-waste criteria for compost (Section 2.7) and the development of LCA Guidelines (Section 2.16).
- The European Parliament has called for a separate Community legislation on bio-waste, for instance in its position to the proposal of the revised Waste Framework Directive (Section 2.5) and in its resolution on a proposal for the soil framework directive (Section 2.9). This issue was a contested one, and the issue of whether a Biowaste Directive, or similar, should be formulated has remained a live one over the last seven (and more) years.

Previous studies undertaken by a consortium lead by Eunomia (see Section 3.1) and by COWI (see Section 3.2) have already taken the matter through the lens of cost-benefit analysis, and have explored the implications of different possible policy approaches.

The current study follows a call from the Commission, asking to take into account the following factors:

- The situation in the EU-27
- The latest legislative development
- The currently available knowledge on the subject

1.2 Objectives of the current study

The objective of this study is to look into ways of improving the way in which bio-waste is managed in the EU, and to provide an appropriate assessment of policy options, including the environmental, economic and social impacts, as well as prospective risks/opportunities. This work could lay the basis for possible additional measures at Community level in this area.

In particular, the project is expected to contribute to the preparatory stages of the Commission's assessment of the bio-waste management options. The Invitation to Tender (ITT) has identified the following Objectives:

- Provide assistance to Green Paper consultations (including analysis and synthesis of the outcome of stakeholder consultations and workshops);
- Provide analysis and synthesis of other consultations, studies and an overview of the state-of-the-art knowledge as contribution to impact assessment;
- Provide an overall assessment of policy options and their relative merits.

The ITT goes on to state that the study should include an assessment of likely benefits and costs of additional or changed policy measures on the management of bio-waste in the EU (including an obligation of separate collection or recycling targets for bio-waste) when compared to the existing policies and those currently being developed. It was expected that the additional measures which would be assessed would include the options already proposed in a preliminary impact assessment ("COWI study" - see Section 3.2), i.e.

- setting compost standards;
- setting compost standards and recycling targets for bio-waste (common to all Member States);
- EU-wide compost standards and recycling targets to be set for individual Member States (or for groups of member states).

The final set of options to be assessed would be based on the results of the Green Paper consultations and the preparatory work undertaken in the context of this study.

During the inception meeting with the Commission services, it was confirmed that a maximum of 4 policy options would be considered.

The final set of options that were withheld for further analysis are described in detail in Chapter 6 of this report.

1.3 Initial approach to this study

The Key Tasks have been set out in the ITT. They are:

- Task 1: Setting up a website in order to support all stages of the work
- Task 2: Provision of services to support stakeholder consultations and workshop related to the Green Paper
- Task 3: Setting and analysing a baseline scenario on bio-waste management in the EU
- Task 4: Assessment of different policy options
- Task 5: Comparison of selected policy options
- Task 6: Consultations, Final Workshop and draft final report

Following the inception meeting and further communication with the Commission services, it was decided to slightly adapt the approach that was initially proposed. We briefly summarize the main steps that were agreed.

1.3.1 Task 1

It was agreed that CIRCA would be used for the stakeholder consultation related to the Green Paper.

1.3.2 Task 2

The Commission had planned to organise a stakeholder meeting to present to stakeholders and wide public the results of the Green Paper consultations. This meeting was initially planned in April/May 2009 but has been replaced by a two-day conference in June 2009. It was agreed that the project team would finalise the analysis of the stakeholders' comments to the Green Paper by then and would present its findings at the conference.

1.3.3 Task 3

The ITT require the development of a baseline scenario by extrapolating the current situation and developments on the EU level and in the Member States by 2020, assuming that no additional specific measures will be taken on bio-waste.

Taking into account the changes in timing for Task 2, it was decided not to wait until the finalisation of Task 2 before proceeding with Task 3. As an alternative approach, it was agreed that the project team would construct a rough baseline based upon publicly available sources and information provided by the Commission, assuming implementation of relevant legislation; if no country-specific information is available, the project team would extrapolate from countries that have published data. This baseline would assume compliance with the Directives (see Section 2) and national legislation and plans. After finalisation, the baseline would be submitted for comments to the MS, to the Council of European Municipalities and Regions, to the FEAD, to the BIR, to ACR+ and to the EEB. The feedback received in this process would be used to improve the baseline. The stakeholders would be given until the 1st of June to comment. In the absence of further comments, the project team would assume that the proposed baseline had been endorsed by the stakeholders.

1.3.4 Task 4 and 5

It was confirmed that the number of policy options to investigate would be limited to four, probably including the three scenarios of the COWI study unless the consultation on the Green Paper would suggest otherwise.

The list of scenarios would be chosen after the finalisation of the Green Paper consultation. Replacing less interesting scenarios by more appropriate scenarios should take place at the same moment.

The final set of options that were withheld for further analysis are described in detail in Chapter 6 of this report.

1.4 The scope of the study

The scope of the study, as set out in the ITT, includes a range of waste streams. It states that:

as a minimum 3 streams should be assessed: waste from agriculture and food industry, green waste and kitchen waste

This is a major departure from previous studies (see Section 3), which have not addressed the issue of 'waste from agriculture and food industry'.

It was agreed during the inception meeting with the Commission services that the emphasis of this project would be on kitchen and green waste streams, taking into account that:

- Information on wastes from agriculture are not always kept by Member States and may have to be estimated from (what may be uncertain) livestock numbers – we refer here to the results from the "Pilot studies on waste from agriculture, forestry and fishing"[2] and COM(2005) 223 final[3]. Moreover, the Green Paper has explicitly excluded forestry or agricultural residues from its scope.
- Information on waste from the food industry is generally scarce and/or of low quality. For waste from the food industry, it was decided that the project team would extrapolate whatever information is available. However, it has turned out during the data collection that almost no member states report systematically on waste arising and even less information is available on treatment. Time series are completely lacking (see Section 4.11 of this report for more details). Therefore, the project team has concluded that an extrapolation in time and space would be so arbitrary as to be meaningless. We have used a case study based approach instead – see Annex D.

Finally, it should be noted that the definition of bio-waste used in the Waste Framework Directive does not include paper, cardboard, nappies and textiles. Therefore, the streams will only be considered inasmuch as they affect the recycling of the waste streams that are included in the definition (see the issues mentioned in Section 2.8 for instance).

The following technology options for biowaste management have not been considered in the study: pyrolysis, gasification, and food waste disposers.

1.5 Structure of this report

Chapter 2 gives an overview of the most important European legislation and policies that affect the management of bio-waste. For the purposes of this study, the following issues have been especially important:

- The targets of the Landfill Directive regarding the reduction of biodegradable municipal waste (*not* biowaste) going to landfill;
- The emission limit values of the Waste Incineration Directive;
- The provisions on bio-waste and on end-of-waste criteria in the Waste Framework Directive;
- The Nitrate Directive, which imposes limits on N loads on farmlands, which can affect the application of compost to land;
- The EU policy for renewable energy, which affects the incentives for the use of bio-waste as a renewable energy resource.

Previous studies, such as those undertaken by Eunomia and COWI, have already taken the matter through the lens of cost-benefit analysis, and have explored the implications of

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[2] See: http://circa.europa.eu/Public/irc/dsis/pip/library?l=/wastesstatisticssregulat/pilotsstudies/wastesfromsagricultures f/management_nace&vm=detailed&sb=Title

[3] "REPORT FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT AND THE COUNCIL on the progress of the pilot studies referred to in Article 4(3) and Article 5(1) of Regulation (EC) No 2150/2002 of the European Parliament and of the Council of 25 November 2002 on waste statistics"

different possible policy approaches. Chapter 3 summarizes the results of these studies and the distinguishing features of our contribution.

Chapter 4 gives an overview of the approach taken to data collection, and discusses the most important problems encountered.

For each country, we have drafted a “fact sheet”, containing all the relevant information regarding biowaste management we have identified, based upon official sources on the one hand and secondary data on the other hand. For the construction of the baseline, a choice between conflicting figures had to be made, often based upon expert judgement from the project team. The figures that were eventually used for the construction of the baseline are always reported.

Chapter 5 outlines the methodology we have used for our baseline construction. This baseline involves a country per country projection until 2020 of biowaste generation on the one hand, and treatment methods on the other hand.

Chapter 6 gives an overview of the policy scenarios that the Commission services have asked us to simulate, also highlighting underlying principles and common assumptions. The following scenarios have been withheld for analysis:

- Scenario 1: compost standard as stand-alone policy instrument:
- Scenario 2: a “high prevention and recycling” scenario; in a variant “scenario 2a” it is assumed that all food waste that is recycled on top of what is recycled in the baseline, is sent to the waste management option that leads to the largest benefits in terms of GHG reduction;
- Scenario 3: a “low recycling” scenario; in a variant “scenario 3a” it is assumed that all food waste that is recycled on top of what is recycled in the baseline, is sent to the waste management option that leads to the largest benefits in terms of GHG reduction;

The objective of Chapter 7 is to understand some of the underlying factors that influence the costs and the benefits of different approaches to biowaste management. The methodology followed, therefore, is to model, for individual Member States, ‘tonne for tonne’ comparisons for the different approaches to biowaste management, highlighting the financial costs and the environmental costs and benefits, and the reasons for variation therein.

Chapters 8 discusses Scenario 1 in more detail. We explain why it is unlikely that such a rapid development of source separation and composting as described in scenario 2 and 3 could materialise without standards and Quality Assurance Systems being developed in the countries concerned. The swift development of a standard might even be a pre-condition for the baselines to be achieved.

Chapters 9 and 10 present and discuss the policy scenarios 2 and 3 in detail. They following points are tackled:

- The waste movements in each scenario
- The financial and economic costs of each scenario
- The changes in greenhouse gas emissions under each scenario

Chapter 11 summarizes. It also compared the GHG reductions estimated under each scenario with the Community’s commitments in this field.

The main report is accompanied by a series of Annexes. The description of the baselines for individual countries is given in Annex A to the report. Besides a detailed description of the methodology used to construct policy scenarios 2 and 3 respectively, Annex B and C report detailed results on a country per country basis.

Annex D provides a separate discussion of industrial bio-waste. Annex E discusses the approach used to estimating the financial costs and benefits of each waste management option, and Annex F gives an overview of the environmental assumptions.

2 Relevant European legislation and policies

2.1 Landfill Directive

Article 5 of the Landfill Directive[4] states that Member States should set up a national strategy for the implementation of the reduction of biodegradable waste going to landfills by means of recycling, composting, biogas production or materials/energy recovery. This strategy should ensure that not later than five years after the date of implementation biodegradable municipal waste going to landfills must be reduced to 75 % of the total amount of biodegradable municipal waste produced *in 1995*. After eight years this must be reduced to 50 % of this amount, and after 15 years to 35 %. Member States that landfilled more than 80 % of their collected municipal waste in 1995 may postpone the attainment of the targets by a period not exceeding four years.

Municipal waste is defined in the Landfill Directive (1999/31/EC) as "waste from households, as well as other waste which, because of its nature or composition, is similar to waste from household". However, the precise definition of biodegradable municipal waste varies from Member State to Member State.

The main motivation for these targets and measures was to reduce the production of methane gas from landfills, inter alia, in order to reduce global warming. And they should also aim at encouraging the separate collection of biodegradable waste, sorting in general, recovery and recycling.

The Report from the Commission on the national strategies for the reduction of biodegradable waste going to landfills[5] points out that all the strategies promote composting, recycling of paper and energy recovery. Most strategies stress the importance of using source segregated organic waste to obtain good quality compost.

At the time the Report was published, the level of detail of the strategies and the measures to achieve the targets varied considerably. Some Member States had chosen legally binding measures, whilst others had chosen voluntary measures and incentives. It was not possible to tell with any certainty from studying the strategies whether the reduction objectives would be met in those Member States which had not yet done so. We will come back to this issue in Section 5.1.

2.2 Incineration Directive

The Waste Incineration Directive (WI Directive) repealed former directives on the incineration of hazardous waste (Directive 94/67/EC) and household waste (Directives 89/369/EEC and 89/429/EEC) and replaced them with a single text.

The Directive sets emission limit values and monitoring requirements for pollutants to air such as dust, nitrogen oxides (NOx), sulphur dioxide (SO_2), hydrogen chloride (HCl), hydrogen fluoride (HF), heavy metals and dioxins and furans. The Directive also sets controls on releases to water in order to reduce the pollution impact of waste incineration and co-incineration on marine and fresh water ecosystems.

[4] Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste (OJ L 182, 16.7.1999, p. 1).

[5] Report from the Commission of 30 March 2005 on the national strategies for the reduction of biodegradable waste going to landfills pursuant to Article 5(1) of Directive 1999/31/EC on the landfill of waste [COM(2005) 105 - not published in the Official Journal].

Most types of waste incineration plants fall within the scope of the Directive, with some exceptions, such as those treating only biomass (e.g. vegetable waste from agriculture and forestry).

Many of the plants that are covered by the WI Directive are also covered by the Integrated Pollution Prevention and Control (IPPC) Directive. In these cases, the WI Directive only sets minimum obligations which are not necessarily sufficient to comply with the IPPC Directive. Such compliance may involve more stringent emission limit values, emission limit values for other substances and other media, and other appropriate conditions.

2.3 Regulation on Animal By-Products

The Regulation on Animal By-Products[6] (ABPR) constitutes the cornerstone of European legislation on food safety.

Animal by-products are defined as the entire bodies or parts of bodies of animals or products of animal origin not intended for human consumption, including ova, embryos and sperm. These materials are then disposed of or processed and re-used in the cosmetics or pharmaceuticals sectors and for other technical purposes.

Following the food crises of the 1990s, such as the bovine spongiform encephalopathy (BSE) epidemic, the role of these by-products in propagating transmissible animal diseases was brought to light. It was concluded that products derived from animals declared unfit for human consumption must not enter the food chain. Moreover, the administration to any animal of proteins obtained by processing carcasses of the same species - or cannibalism - may constitute an additional risk of disease propagation.

This Regulation sets out the measures to be implemented for the processing of animal by-products (including their collection, storage, and transport). Laying down minimum rules at European level, it gives the Member States the option of taking even more restrictive measures or measures covering products excluded from its scope.

Regulation (EC) 1774/2002 prohibits many animal by-products (ABP) from being disposed of directly to landfill. The Regulation categorises ABP into three categories, according to risk:

Category 1 - very high risk, i.e. animals suspected or confirmed as being infected by BSE (Bovine Spongiform Encephalopathy);

Category 2 - high risk, i.e. condemned meat, fallen stock, manure, digestive tract content; and

Category 3 - low risk, i.e. ‘catering’ wastes, former foodstuffs and raw meat/fish from food manufacturers and food retailers.

The APBR does not apply to catering waste, unless it is destined for animal consumption, it is destined for use in a biogas plant or for composting or it comes from means of transport operating internationally. Catering waste from means of transport operating internationally is part of Category 1 waste.

According to Article 7, animal by-products and processed products, with the exception of Category 3 catering waste[7] shall be collected, transported and identified in accordance with Annex II of the Regulation.

[6] Regulation (EC) No 1774/2002 of the European Parliament and of the Council of 3 October 2002 laying down health rules concerning animal by-products not intended for human consumption (OJ L 273, 10.10.2002, p. 1).

[7] Category 3 catering waste is catering waste that does not originate from means of transport operating internationally.

However, Article 4 also requires Member States to take the necessary measures to ensure that Category 3 catering waste is collected, transported and disposed of without endangering human health and without harming the environment. According to Article 6, Category 3 catering waste shall be transformed in a biogas plant or composted. It is prohibited to feed farmed animals other than fur animals with catering waste.

Thus catering waste may be processed in accordance with national law until the Commission determines harmonised measures following the comitology procedure described in Art. 33(2) ABPR. As no harmonised process requirements have as yet been proposed by the Commission, the Member States can still regulate the treatment of catering waste in compost and biogas plants.

In April 2004 the Commission published the “Guidance on applying the new Animal By-Products Regulation (EC) No 1774/2002” where it clarifies that the regulation abstains from a detailed provision for catering waste in favour of foreseen environmental legislation or national rules[8].

According to Barth et al. (2008)[9] , many Member States up to now misinterpreted the possibility to introduce more relaxed rules for composting of catering waste at least from source separated organic household waste and have taken over the full set of requirements of Annex VI of the ABPR in national licensing and plant approvals. Annex 3 of Barth et al. contains an overview of how *some* Member States have implemented the ABPR.

2.4 Thematic Strategy on the Prevention and Recycling of Waste

The Thematic Strategy on the Prevention and Recycling of Waste[10] refers to the report on national strategies[11], and points out that there is no single environmentally best option for the management of biowaste that is diverted from landfill. It concludes that management for this type of waste should be determined by the Member States using life-cycle thinking.

It expressed the intention to produce guidelines on applying life-cycle thinking to the management of biowaste, to communicate these guidelines to Member States and to invite them to revisit their national strategies.

It also announced the adoption of compost quality criteria under the end-of-waste provision proposed for the Waste Framework Directive and to bring the biological treatment of waste under the scope of the IPPC Directive when it is revised.

Finally, it foresees a revision of Council Directive 86/278/EEC on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture.

All these elements are further discussed below.

2.5 Waste Framework Directive

The Waste Framework Directive 2006/12/EC has been revised. On 17 June 2008, the European Parliament adopted a legislative resolution in which it approved the Council's

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[8] ec.europa.eu/food/food/biosafety/animalbyproducts/guidance_faq_en.pdf

[9] Barth et al (2008), Compost production and use in the EU, report to the European Commission, Joint Research Centre/ITPS, Final report

[10] Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and The Committee of the Regions - Taking sustainable use of resources forward - A Thematic Strategy on the prevention and recycling of waste COM(2005) 666 final

[11] Report from the Commission of 30 March 2005 on the national strategies for the reduction of biodegradable waste going to landfills pursuant to Article 5(1) of Directive 1999/31/EC on the landfill of waste [COM(2005) 105 - not published in the Official Journal].

Common Position as amended. This step marks the adoption of the revised Directive in second reading and the end of the negotiations.

The new Waste Framework Directive foresees in its article 22 specific provisions on bio-waste. Member States are obliged , as appropriate, to encourage the treatment of bio-waste following the waste treatment hierarchy by promoting separate collection with a view to the composting and digestion of bio-waste, by taking measures for the treatment of bio-waste in a way that fulfils a high level of environmental protection, and by stimulating the use of environmentally safe materials (e.g. composts) produced from bio-waste.

In a crucial clause, *the Commission is asked to carry out an assessment on the management of bio-waste* with a view to submitting a proposal if appropriate. In this assessment the opportunity should be examined of setting minimum requirements for bio-waste management and quality criteria for compost and digestate from bio-waste. It is envisaged that this could end up in a Communication or in a specific bio-waste Directive or Regulation. It is clear that this constitutes the point of departure for this study.

Article 11 introduces reuse and recycling targets. Bio-waste however is not included in the waste types that are to be collected separately or for which recycling targets have been established. However, Member States are allowed and encouraged to include more waste streams, to promote high quality recycling. To this end they can set up extra separate collection schemes of waste where this is technically, environmentally and economically practicable and appropriate to meet the necessary quality standards for the relevant recycling sectors. By 31 December 2014 at the latest the Commission itself shall examine the existing measures and targets and shall consider setting targets for other waste streams.

The new Waste Framework Directive introduces an important new element on energy recovery through anaerobic digestion of biodegradable waste. Article 2 point 1 f extends the exclusion of “other natural non-hazardous agricultural or forestry material” from the application of the Waste Framework Directive. In the old Waste Framework Directive 2006/12/EC this was limited to application of this waste for use in farming. The new Waste Framework Directive foresees an exclusion for “the production of energy from such biomass”. This means that installations for composting for this material do fall under the restrictions and obligations of the environmental permit for recycling activities while competing installations for bio-methanisation and energy recovery are exempted.

Art 6 specifies that certain specified waste shall cease to be waste when it has undergone a recovery, including recycling, operation and complies with specific criteria to be developed in accordance with the following conditions:

- the substance or object is commonly used for specific purposes ;
- a market or demand exists for such a substance or object;
- the substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products; and
- the use of the substance or object will not lead to overall adverse environmental or human health impacts.

The measures relating to the adoption of such criteria and specifying the waste shall be adopted using the comitology procedure. End-of-waste specific criteria should be considered, among others, at least for aggregates, paper, glass, metal, tyres and textiles.

Where criteria have not been set at Community level, Member States may decide case by case whether certain waste has ceased to be waste taking into account the applicable case law.

The Joint Research Centre (JRC) is working on a project to look at the scientific methodology that could be used to determine end of waste criteria, including for compost – see Section 2.7.

2.6 Revision of the IPPC Directive

On 21 December 2007 the Commission adopted a Proposal for a Directive on industrial emissions[12]. The Proposal recasts seven existing Directives related to industrial emissions into a single legislative instrument. The recast includes in particular the IPPC Directive and the WI Directive.

The IPPC Directive[13] established a set of common rules for permitting and controlling industrial installations - it has recently been codified (Directive 2008/1/EC).

Operators of industrial installations covered by Annex I of the IPPC Directive are required to obtain an authorisation (environmental permit) from the authorities in the EU countries. New installations, and existing installations which are subject to "substantial changes", have been required to meet the requirements of the IPPC Directive since 30 October 1999. Other existing installations had to be brought into compliance by 30 October 2007.

The IPPC Directive is based on the following principles:

The integrated approach means that the permits must take into account the whole environmental performance of the plant.

The permit conditions including emission limit values (ELVs) must be based on Best Available Techniques (BAT), as defined in the IPPC Directive. To assist the licensing authorities and companies to determine BAT, the Commission organises an exchange of information between experts from the EU Member States, industry and environmental organisations. This results in the adoption and publication by the Commission of the BAT Reference Documents (the so-called BREFs).

The IPPC Directive contains elements of flexibility by allowing the licensing authorities, in determining permit conditions, to take into account: (a) the technical characteristics of the installation, (b) its geographical location and (c) the local environmental conditions.

The Directive ensures that the public has a right to participate in the decision making process, and to be informed of its consequences

The Impact Assessment for the proposed Directive on industrial emissions identified inconsistencies related to the biological treatment of organic waste and recommended including this sector in the IPPC Directive[14]. It pointed out that this type of waste treatment is covered under the current scope of the IPPC Directive only if it results in final compounds or mixtures which are discarded through disposal operations. The relevant BREFs contain BAT conclusions for these types of installations. This means that similar installations (with similar environmental impacts) resulting in waste or products (eg composting) which are not disposed of but recovered or used as products are not covered under the scope of the IPPC Directive. According to the Impact Assessment undertaken for this Directive, these inconsistencies result in possible distortion of

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[12] Proposal for a Directive of the European Parliament and of the Council on industrial emissions (integrated pollution prevention and control) (Recast) [COM(2007) 843 final] [SEC(2007) 1679] [SEC(2007) 1682].

[13] Directive 1996/61/EC

[14] http://ec.europa.eu/governance/impact/docs/ia_2007/sec_2007_1679_en.pdf

competition between similar types of installations and a lower level of environmental protection for installations not covered under the IPPC Directive.

The IA argues that the economic impacts of BAT implementation are limited. The smallest installations would not be covered by the IPPC Directive since their production capacity is below 50 tonnes per day. The IA recommends to cover this sector under the IPPC Directive.

This advice has been followed in the proposal for the Directive.

The opposite view was taken by Barth et al. (2008), who believe that a binding BAT would impose a disproportionate burden upon composting and therefore constitute a significant handicap for the implementation of cost effective and environmentally sound systems in many Member States where biowaste treatment is still in its infancy.

2.7 End-of-waste criteria

As explained in Section 2.5, the JRC is working on a project to look at the scientific methodology that could be used to determine end of waste criteria.

The objectives of the project are to:

- Develop a general methodology for determining end of waste criteria (end of waste methodology) using three specific pilot case studies, including compost
- Methodology should be generally suitable for application to any candidate waste stream to determine (if any) end of waste criteria.
- Propose further candidate waste streams for consideration based on standard selection criteria.

The reports to DG ENV were due in 2008.

The report by Barth et al. (2008) is an essential input in this process, and has been used extensively in this report.

2.8 Packaging Directive

The Packaging Directive (94/62/EC as amended by 2004/12/EC) among other provisions sets minimum recycling targets for paper and board *packaging* waste. Compliance with the Packaging Directive thus directly affects the amounts of biodegradable waste landfilled or incinerated, and thus also compliance with the Landfill Directive. However, it does not affect recycling of bio-waste as defined in the Waste Framework Directive.

In other words, compliance with the Packaging Directive makes it easier to comply with the Landfill Directive without having to increase the amounts of bio-waste that are recycled.

All other things being equal, it can be concluded that the Packaging Directive provides a negative incentive for the recycling of bio-waste as defined in the Waste Framework Directive.

However, the Packaging Directive does provide some positive incentive as well, to the extent that some countries include cardboard packaging within the management of biowaste through composting and anaerobic digestion. For example, some anaerobic digestion plants treat a waste stream which includes dirty card, whilst some composting plants treat card which is collected alongside biowaste. If there are increasing returns to scale in biowaste treatment, this lowers the average costs. This is not, however, ‘mainstream activity’, either for card or for biowaste management. As such, we do not intend to focus on this in the study.

2.9 EU soil strategy

The Commission adopted a Soil Thematic Strategy (COM(2006) 231) and a proposal for a Soil Framework Directive (COM(2006) 232) on 22 September 2006 with the objective to protect soils across the EU.

The draft Soil Framework Directive imposes the obligation for member States to design programmes of measures to combat organic matter decline (Article 8). Member States are requested to draw up, at the appropriate level, a programme of measures including at least risk reduction targets, the appropriate measures for reaching those targets, a timetable for the implementation of those measures and an estimate of the allocation of private or public means for the funding of those measures.

In Article 10 of its resolution on this proposal[15], the European Parliament requires that "the Commission shall present a proposal for a biowaste Directive setting quality standards for the use of biowaste as a soil improver."

This requirement can be seen in the light of a trade-off identified in the EU Soil strategy: on the one hand, compost is identified as a tool to fight the decline of organic matter in soils (see Section 4.1.1 of the Communication); on the other hand, there is the need to prevent soil contamination (see Section 4.1.2 of the Communication).

2.10 European Climate Change Programme

The European Climate Change Programme considers promoting organic input on arable land (crop residues, cover crops, farm yard manure, compost, sewage sludge) as a tool to reduce Greenhouse gas emissions[16].

2.11 Soil protection when sewage sludge is used

Currently, the use of sewage sludge is governed by Council Directive 86/278/EEC of 12 June 1986 (amended by Council Directive 91/692/EEC and Council Regulation (EC) No 807/2003 of 14 April 2003) -, sewage sludge may be used in agriculture provided that the Member State concerned regulates its use. The Directive lays down limit values for concentrations of heavy metals in the soil (Annex IA), in sludge (Annex IB) and for the maximum annual quantities of heavy metals which may be introduced into the soil (Annex IC). Sludge must be treated before being used in agriculture but the Member States may authorise the use of untreated sludge if it is injected or worked into the soil (Article 6). The use of sludge is prohibited on some types of crops or in grounds intended for the cultivation of some specific types of crops (Article 7). Member States may take more stringent measures than those provided for in this Directive (Article 12).

The Thematic Strategy on the Prevention and Recycling of Waste foresees a revision of this Directive.

2.12 Nitrate Directive

Council Directive 91/676/EEC, the Nitrate Directive, imposes limits on N loads on farmlands.

Barth et al. (2008) point out that this in general may impose a constraint on the use of soil improvers, but may also trigger a greater application of compost as a replacement of

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[15] European Parliament legislative resolution of 14 November 2007 on the proposal for a Directive of the European Parliament and of the Council establishing a framework for the protection of soil and amending Directive 2004/35/EC (COM(2006)0232 – C6-0307/2006 – 2006/0086(COD)) - P6_TA(2007)0509 – art. 8a (amendment 66)

[16] http://ec.europa.eu/environment/climat/pdf/finalreport_agricsoils.pdf

mineral fertilisers, given the lower N availability and the fact that compost is a slow-release source of N. Some EU Member States have already enforced related provisions that recognise such an important feature of compost, thereby driving a higher application of it instead of liquid slurries or mineral fertilisers.

2.13 Biofuels Directive

Directive 2003/30/EC (the "Biofuels Directive") entered into force in May 2003, and stipulates that national measures must be taken by countries across the EU aiming at replacing 5.75 % (calculated on the basis of energy content) of all transport fossil fuels (petrol and diesel) with biofuels by 2010. According to the Directive, "biofuels" shall mean liquid or gaseous fuel for transport produced from biomass, where biomass means the biodegradable fraction of products, waste and residues from agriculture (including vegetable and animal substances), forestry and related industries, as well as the biodegradable fraction of industrial and municipal waste.

As the biogas resulting from anaerobic digestion can be upgraded for use as vehicle fuel (see Annex F to this report), the Biofuels Directive provides, all other things being equal, an incentive for the expansion of AD.

2.14 EU Policy for Renewable Energy and Directive on Renewable Energy Sources

The EU Policy for Renewable Energy and Directive on Renewable Energy Sources RES 2001/77 (amended by Directive 2004/8/EC on the promotion of cogeneration based on a useful heat demand in the internal energy market and by Directive 2009/28/EC) establishes targets for total Renewable Energy Sources. As Biomass accounts for a relatively large share in total RES, this may lead to competing demands for biomass.

In the accompanying text to the Proposal for Directive 2009/28/EC [17], it was recognized that the negative effects suggested by respondents during the Impact Assessment mostly relate to the pressure on biomass resources, which are also used for non-energy industrial use and its further exploitation may lead to shortages or undesirable environmental impacts. This is reflected in several clauses of the Proposal.

2.15 Common Agricultural Policy

On 26 June 2003, EU farm ministers adopted a fundamental reform of the CAP, based on "decoupling" subsidies from particular crops. The new "single farm payments" are subject to 'cross-compliance' conditions relating to environmental, food safety and animal welfare standards ('cross-compliance')[18]. According to Barth et al (2008, p 174), some countries have included the principles of "humus/organic matter management" in these requirements and check it in the frame of the cross compliance obligations. This might include the use of more compost by the farmers.

2.16 Developments of LCA Guidelines

As explained in Section 2.4, the Commission is preparing guidelines addressed to policy makers on the application of life cycle thinking to biowaste management policies. The Joint Research Centre is assisting DG Environment with this task.

The project includes two major steps:

- Analysis and brief report on existing studies and related expertise

[17] COM(2008) 19 final.
[18] http://ec.europa.eu/agriculture/envir/index_en.htm#crosscom

- Development of guidelines with a supporting tool, documentation and data

The guidelines and supporting tool will be developed with the direct involvement of an experts advisory panel and then submitted to a large stakeholder consultation. Updated information can be found at the JRC website devoted to the European life cycle thinking guidelines for the management of municipal biodegradable waste.

The JRC provides technical and scientific support for integrating life cycle thinking effectively into EU and Member State policies, developing recommended approaches and guidelines, indicators, reference data, and pilot studies to facilitate life cycle thinking in waste management[19].

This includes the development of European life cycle thinking guidelines for the management of biodegradable waste.

According to the JRC, the draft guidelines cannot be shared yet.

A specific mention should be made here of the International Reference Life Cycle Data System (ILCD). The ILCD System addresses the whole range of life cycle related approaches and applications, taking a global perspective. The draft scope of the International Reference Life Cycle Data System (ILCD) covers a set of technical guidance documents, supporting tools and documents, data sets, and other resources in support of good practice in Life Cycle Assessment (LCA). Some of the ILCD components are already available, others are still under development, with key deliverables were expected by early 2009.[20]

[19] http://lca.jrc.ec.europa.eu/waste/
[20] http://lca.jrc.ec.europa.eu/EPLCA/deliverables.htm

3 Previous work on this subject

3.1 Eunomia study

The main objective of the study undertaken by the consortium lead by Eunomia was 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 (MSW). Although all management options (anaerobic digestion, composting, landfilling, incineration, etc.) were considered in the study, much of the emphasis of the study was on the separate collection and recycling of the biodegradable fraction of MSW.

The study concluded that a policy of source separation would be justified where the collection system for source-separated biowastes was carried out in such a way as to optimise costs. Furthermore, the study claims that where the costs of biowaste treatment itself are kept to a reasonable level, it becomes likely that the net private cost increase will be minimal, or negative.

The study argues that the more difficult it becomes to ensure that 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.

According to the authors, 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.

3.2 COWI study

The COWI study "Preliminary Impact Assessment for an Initiative on the Biological Treatment of Biodegradable Waste" (December 2004) aimed "to assess the main social, economic and environmental implications of a selected set of policy options that may be put in place to provide for better and/or increased biological treatment of biodegradable waste". It focused on food and green waste (FGW) in biodegradable municipal waste (BMW).

The COWI study has analysed the following options, where compost standards would be an integral part of the other options (p. 33):

- Mandatory separate collection where common rules are established
- Setting targets for separate collection where the means to comply with those targets may differ
- Compost standards defining EU wide common standards for the quality of end products as well as the treatment process
- Tradable landfill permits which would involve the establishment of tradable permit systems in the individual EU Member States

Tradable permits have only been discussed in general terms. The reports concludes that whether they would be a good instrument depends largely on local conditions and contexts.

Other options that have been considered but omitted include: voluntary agreements, EU wide taxes, awareness raising instruments, public procurement policy and variable charging.

The COWI analysis has assumed that standards relate to the end-product, neither to the feedstocks nor the processes.

The key conclusion of this study was that the Landfill Directive in itself would provide for substantial increases in the amounts of FGW that is biologically treated but that common standards for compost could provide a positive additional contribution to this development. Mandatory separate collection[21] that translates into 85% of the FGW being biologically treated would provide an additional increase in the amounts of biologically treated FWG in most countries (that do not deliver this high fraction in the Baseline Scenario). The modest size of the net benefit reported is to a large extent attributable to the relatively small change that the mandatory separate collection will involve compared to the Baseline Scenario.

According to the COWI study, the setting of national targets[22] can provide a further positive contribution. Such national targets could ensure that a certain fraction of the FWG will be biologically treated and could be an important factor in accelerating the process towards compliance with the Landfill Directive.

In relation to the compost market, the COWI study points out that the standards can assist to build up consumers' confidence in the product, while the targets ensure that sufficient quantities of sorted waste are generated thereby justifying investments in the necessary composting facilities. Compared to mandatory separate collection, the setting of targets would have the advantage of maintaining a certain level of freedom of action for individual Member States in framing their specific strategy for compliance.

For all three countries covered in the case studies (Portugal, Ireland and Sweden) composting is preferable to incineration. Change from landfill to composting is expected to result in a net reduction of total costs because reductions in treatment costs and external costs more than outweigh the additional collection costs. Furthermore, switch from landfill to incineration is expected to result in a significant increase in the total costs due to both higher treatment costs and higher external costs.

From a pure environmental point of view, the COWI study concludes that a switch from landfill and incineration is favourable to both composting and anaerobic digestion.

Further, according to COWI, it seems that in general from an economic point of view switches to composting are more advantageous than switches to anaerobic digestion. This is because the treatment costs are much higher for anaerobic digestion than for composting outweighing that the environmental benefits are generally higher for anaerobic digestion than for composting.

3.3 Distinguishing features of this study

The subject of the current study is at the same time broader and narrower than the subject of the Eunomia and the COWI study. It is narrower because it covers only bio-waste, which is a fraction of biodegradable waste (which also includes, for instance, paper and paperboard); it is broader because it is not limited *a priori* to municipal waste. We shall see in Section 4 that this departure from the conventional scope of this type of study is severely limited by the availability of data.

Another important change of scope is that the current study covers the EU27, rather than the EU15. This brings complications that go beyond the simple increase in the number of

[21] Compost standards would be an integral part of this option.

[22] Targets in the COWI study may relate to the amount of biodegradable waste that must be biologically treated, or to the fractions of biodegradable waste that must be separately collected. Compost standards would be an integral part of this option as well.

cases that need to be covered. First, data are even scarcer in the EU12 than in the EU15. Second, existing waste management practices in the EU12 differ sometimes significantly from the practices in the EU15. This is due to a series of factors: inherited (lack of) infrastructure; different relative costs of labour, land and capital (which affects the relative attractiveness of different waste collection and treatment options); different priorities in environmental policy etc. However, fewer costs have been sunk (in for instance incineration capacity).

This study has also to take into account new legislative developments such as: the Thematic Strategy on the Prevention and Recycling of Waste, the Waste Framework Directive, the Directive on the promotion of the use of renewable energy…

Finally, this study will use the most recent estimates of the private and social costs of the different waste treatment options.

4 Data collection

This section provides an overview of the approach that was taken for the collection of the data needed for the baseline scenario, and assesses the limitations of the data that have been used.

4.1 General approach

The official reporting requirements with respect to the subject of this study are rather limited.

Member States have to submit the following information to the Commission:

- their waste management plan (according to the Waste Framework Directive)
- their national strategies for the implementation of the reduction of biodegradable waste going to landfills (according to the Landfill Directive)

Whenever these strategies have been put in the public domain, and have been made available in English or in French, they have been used as an input on this study. However, this situation was rather exceptional. As the report from the Commission on the national strategies[23] dates back from 2005, it does not cover the EU12.

Therefore, the project team has had to rely mainly on secondary sources.

The most important secondary sources were the Country Fact Sheets of the European Topic Centre on Sustainable Consumption and Production ETC/SCP (previously European Topic Centre on Resource and Waste Management).[24] These fact sheets contain in general some information on how the European Directives have been transposed into national law, and report data on the amounts of MSW and BMW generated and sent to landfill. However, most information that was available when the baseline scenarios were constructed is old (2004 and earlier).

Besides the Country Fact Sheets, we have systematically verified the information provided by the European Environment Agency (EEA).

These data sources have been complemented with a variety of other data sources, mainly reports from national environment ministries, waste management agencies and statistical agencies.

For each country, we have drafted a “fact sheet”, containing all the information we have identified. A detailed analysis of the individual “fact sheets” shows that, depending on the data source, some figures on the same subject vary widely. In some countries, there is a very significant discrepancy between some of the figures provided by the ETC/SCP and data from national sources. However, even national sources are not always consistent.

We will discuss some of the recurring inconsistencies and uncertainties below. For the construction of the baseline, a choice between conflicting figures had to be made, often based upon expert judgement from the project team. The figures that were eventually used for the construction of the baseline are always reported.

[23] COM(2005) final.

[24] http://scp.eionet.europa.eu/facts/factsheets_waste .

4.2 Economic and demographic projection

For our forward projections, we have used the “European Energy and Transport - Trends to 2030. Update 2007” published by DG TREN. This way, we are consistent with the scenarios used by the Commission for its own long term forecasts in the fields of transport and energy. Our assumption is thus that the current economic crisis does not affect long term prospects. We think this is a reasonable approach for most European countries[25]. Long term economic growth is fundamentally determined by supply-side factors (demography and total factor productivity growth), while the current recession is essentially a demand-side crisis. Even if strong negative growth is observed in 2008-2009, a downturn in the business cycle should not affect long-term perspectives when looking at a 2020 horizon.

4.3 Waste treatment capacities

EUROSTAT data are available on existing waste treatment capacities, but they are insufficiently detailed to be of much use for the purposes of this study: for instance, it is not specified which part of incineration capacity is destined for MSW.

Even when national data on composting plants are available, the specific option used (windrow composting, in-vessel composting…) is often not specified, even though environmental impacts such as odour nuisance vary significantly according to the treatment method.

Next to nothing is reported on the distribution of size classes, which adds a further element of uncertainty if treatment costs are subject to (dis)economies of scale.

4.4 Price information

The Confederation of European Waste-to-Energy Plants (CEWEP) provides some information on existing gate fees. No other EU-wide information on gate fees and waste related taxes and prices has been identified in the public domain.

When national data has been identified, they have been reported in the “country sheet”. It is in general not specified:

- whether the prices include taxes or not
- whether they are indexed

4.5 Cost of treatment methods

With the exception of France and The Netherlands, no country specific data on the costs of different waste treatment methods have been identified. Some data on collection costs are available from Flanders, France and Portugal.

Some data are available from international sources. The main drawback of utilising these sources is of course that they do not allow taking into account local circumstances.

The JRC report "Environmental assessment of municipal waste management scenarios Part II" provides unit costs for:

- Kerb-side collection
- Costs of bring systems

[25] Possible exceptions could be some countries that have been particularly hard hit by a bust in the construction sector or who are currently recovering from string external imbalances. Contrary to the US economy, the European economy does not suffer from high levels of consumer indebtedness, which could lead to weak growth in consumer spending in the upcoming years.

- Directive compliant landfills (depending on capacities)
- Incineration
- Composting (both windrow and AD)

The IEA[26] provides investment costs and *energy* production costs for MSW incineration, biogas digestion (including from landfills).

Murphy and McKeogh[27] provide unit costs for incineration, gasification, AD plants (differentiated according to capacity and biogas use).

Last[28] provides estimates of land requirements, capital costs and operating costs for: MBT plants, incineration plants, AD plants, windrow composting processes, in-vessel composting processes, gasification plants.

AWARENET provides costs for incineration, gasification-pyrolysis, composting with forced aeration, composting without forced aeration, anaerobic separate digestion, anaerobic co-digestion, biodiesel[29].

4.6 Bio-waste arisings

Taking into account that the targets of the Landfill Directive refer to biodegradable municipal waste (BMW), it is not surprising that publicly available data often refer to BMW arisings, and not to bio-waste. In the absence of other data, assumptions need to be made with respect to the share of BMW that falls within the definition of bio-waste (see Section 5.4.3.3 for details on how we have proceeded).

Some member states report other fractions than BMW. It is not always clear how these fractions are defined. Moreover, even when they are defined, the definitions are often highly idiosyncratic, and it is not always clear how these fractions are related to bio-waste as understood by the Green Paper. We give here some examples of countries that have provided such information:

- Austria reports separately on the following categories: green waste; market waste; kitchen and canteen waste; food, beverage and tobacco waste; waste from vegetable and animal fat products; other waste from the processing and refinement of animal and vegetable products. Moreover, each stream is defined explicitly.
- The Region of Flanders in Belgium distinguishes the following categories of bio-waste: green waste; vegetable, fruit and garden waste (VFG) and industrial organic waste.
- Cyprus reports on “organic” waste rather than on bio-waste.
- The Danish Environmental Protection Agency distinguishes: service sector, industry and domestic waste.
- Estonia reports on how BMW is split up in: kitchen waste, paper and cardbaord waste, garden waste and wood waste. However, the quantities reported are based upon a sampling that took place in 2005. It is assumed that the proportions have remained constant since 2000; actual composition may thus be different from the quantities reported.

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[26] International Energy Agency (2008), Deploying Renewables. Principles for Effective Policies.

[27] Murphy, J.D. and McKeogh; E. (2004), Technical, economic and environmental analysis of energy production from municipal solid waste, Renewable Energy 29, pp 1043-1057.

[28] Last, S (2008), An Introduction to Waste Technologies, The processes Used to Recycle, Treat, and Divert Municipal Solid Waste Away from Landfills, Waste Technologies UK Associates.

[29] AWARENET, Agro-food waste minimisation and reduction network, Handbook for the prevention and minimisation of waste and valorisation of by-products in European agro-food industries, funded by the Growth Programme, European Commission, Project N° GRD1-CT2000-28033.

- France reports on biodegradable waste, split up as follows: biodegradable household waste, garden waste and non-household paper/paperboard packaging.
- Ireland splits up BMW in: wood, paper and cardboard, municipal organic waste (composed of household and commercial organic waste) and textiles. Organic waste is food and garden waste.
- The Netherlands report on kitchen and green waste in household waste on the one hand and collected separately on the other hand.
- For 2004, Poland reports data on: paper and cardboard collected separately; cloth and textiles (made of natural materials) collected in a selective manner; garden and park wastes; biodegradable waste coming into stream of mixed municipal waste; waste from market places (biodegradable proportion thereof).
- Portugal splits BMW in “paper and cardboard” and “fermentable products”.
- Slovenia reports on “biodegradable garden and park waste” and “biodegradable kitchen and canteen waste”.
- Barth et al. (2008) define biowaste as a mixture of kitchen and garden waste from source separated collection of organic household waste. This is the material commonly collected in the commingled collection scheme for food and garden waste (brown bin, biobin system).

It is clear that most of these data are not immediately comparable.

Additional uncertainty with respect to composition of household waste is due to the fact that several MS do not provide data on the organisation and coverage of separate collection. Moreover, the collection and treatment of kitchen waste depends on the implementation of the ABPR, which can vary even *within* countries (see Section 2.3).

Finally, it is not always clear how reported quantities have been estimated.

In some cases, it is clear that they are **not** the result of any actual measurement in the report year. For instance, in Cyprus, data on waste generation for the years 1995-2001 are estimates made using linear interpolation and taking into account the resident population and the number of tourists visiting the country.

In other countries, there are some doubts on the validity of official data. For instance, in Bulgaria, local experts reckon that the official estimates exaggerate waste generation.

Changes in the estimation methods also occur trough time. In Estonia, this has led to a drop in the estimates of waste generation by about one third in 2001 compared to 2000. Because Estonia is transparent on this change in the estimation method, the abrupt change in the time series can be understood. However, in some other cases, time series exhibit some fluctuations that are difficult to grasp.

We can conclude that any data on bio-waste has to be treated with a lot of circumspection. A lot of uncertainty surrounds the estimates that are used in this study, and virtually nothing is known on the margins of error in the estimates provided by the Member States.

4.7 Waste treatment and disposal

Insofar as information is available on the different treatment and disposal methods, the composition of the quantities processed per method is, in general, not reported.

Some exceptions are:

- Austria provides figures on the origin of landfilled waste from households and similar establishments (direct and after splitting of residual and bulky waste; incineration ashes; from MBT; after sorting of seperately collected waste; after sorting of biogenic waste); on the origin of incinerated municipal waste (residual and bulky waste; from MBT; from separate collection).

- In Denmark, the Waste Statistics records the quantities of organic waste (from either municipal solid waste or other sources) which is anaerobically digested, composted or used for other purposes such as animal feed.
- Ireland provides figures on the origin of landfilled BMW, following the categories described in Section 4.6.
- Italy splits the waste quantities composted up in “urban household organic waste from source separated collection” and “green or garden waste”. The composition of the first category is not specified however.
- Latvia and Lithuania provide a detailed breakdown of the amounts of BMW and other biodegradable waste going to landfills.
- Poland reports the amounts of biodegradable waste sent to landfill split up according to the European waste catalogue. Poland also provides data on the amount of waste treated biologically, but without specifying the method (compost, AD, MBT).
- Poland also publishes data on the recovery and disposal of waste from agriculture, horticulture, aquaculture, forestry, hunting and fishing, and food preparation and processing. However, the recovery or disposal method is not split according to the origin of the waste.

Another complication is that the categories are not always explicitly defined. For instance, in France, data is available on “green waste composting”, “biological waste composting” and “household waste composting”. It is not clear what should be understood by “biological waste” in this context.

Very little information is provided on what happens with residues of incineration. Austria and Denmark are amongst the exceptions.

In countries where anaerobic digestion (AD) is used, the quantities of municipal bio-waste co-fermented in agricultural biogas plants or with sludges are not always known (see for instance Austria). Therefore, even when the quantities of digestate or biogas produced by AD plants are known, it is often difficult or impossible to trace what portion comes from agriculture waste and sludges on the one hand and BMW on the other hand.

Sometimes, recycling and recovery rates are reported, but without mentioning the actual method used. This for instance the case with recovery rates for BMW in Ireland. However, both the environmental impact and the private costs of different recycling techniques can differ widely. Therefore, figures on existing biowaste recycling are not particularly informative for the purposes of this study if they are not split up according to the method used.

4.8 Compost

Barth et al. (2008) provide a comprehensive and recent overview of compost production, use and prices throughout the EU27. The regulatory constraints are reported in detail. Together with the country presentations on the website of the European Compost Network, this is the main source of information for all issues specifically related to compost.

4.9 Energy from waste treatment

The RENEWABLE ENERGY COUNTRY PROFILES drafted as part of the PROGRESS report[30] (February 2008) provide detailed information on support schemes for renewable energy (including biogas and landfill gas) for the EU27. The main limitation of this data source is that it is not always clear whether the reported guaranteed feed-in tariffs for renewable energy are indexed. Also, for some support schemes, *maximum* rates of support (rather than effective rates) are provided. However, in a personal correspondence, DG TREN has referred to this report as one of the scarce comprehensive sources of information available on this subject. We have certainly not identified a more recent or more complete source.

Energy production from biogas and *renewable* MSW has been obtained from “Systèmes Solaires. Le journal des énergies renouvelables”. July-August 2008. The information covers the EU27 (if national data are available) in a systematic way. An explicit differentiation is made between biogas from landfills, from sewage sludge and from other sources (including anaerobic digestion).

We think this source is more appropriate for our purposes than the EUROSTAT statistics, who report on *total* energy obtained from MSW and biogas (without further differentiation).

4.10 Illegal and informal disposal

In several MS (e.g. Estonia, Ireland, Latvia), municipal waste collection services do not cover the entire population. In these cases, municipal waste is disposed off in illegal (backyard burning, fly-tipping) or informal (home composting) ways. While it is clear that these disposal methods entail negative environmental impacts, their exact magnitude is, per definition, not known.

Information on the amounts of biowaste treated through home composting has only been provided by a limited number of countries (for instance Austria, Estonia, Hungary and Ireland).

4.11 Bio-waste from the food and catering industry

Information on bio-waste from the food industry and from the catering industry has turned out to be even scarcer than expected.

In the absence of any European requirement, no systemic data on this issue is provided by the Member States.

The only comprehensive study we have identified on the subject was provided by the AWARENET project.[31]

AWARENET has concluded that it is not clear what should be understood by “food waste”. A particular problem is the dividing line between waste and by-products – this is especially problematic because the definition used in the Green Paper excludes those by-products of food production that never become waste. AWARENET members have agreed on the percentages of waste and by-products generated for each sub-sector and production process – it is assumed that these percentages are applicable to all MS covered by the study. Food production volumes have been evaluated using the official

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[30] Promotion and Growth of Renewable Energy Sources and Systems. The PROGRESS project was supported by the European Commission, DG Energy and Transport, under contract no. TREN/D1/42-2005/S07.56988.

[31] Agro-food wastes minimisation and reduction network (AWARENET), Handbook for the prevention and minimisation of waste and valorisation of by-products in European agro-food industries, funded by the GROWTH Programme of the European Commission.

data from the European Production and Market Statistics (EUROPROMS) on the one hand and figures coming from individual companies and sectorial associations on the other hand. For the 18 countries covered by the study (EU15, Norway, Poland and Hungary), this has allowed to estimate the actual amounts of waste generated – these amounts are reported in the “country sheets” in annex to the current report. It is estimated that, in those countries, a total of 222 million ton of food waste and by-products are generated per year. According to AWARENET, most of these wastes or by-products go through valorisation stages mainly for animal feed, spread on land, composting or higher added value products. The handbook provides detailed descriptions of the different technologies for food by-product valorisation, but no data are provided on the actual *amounts* of food waste and by-products processed according to each valorisation method. In a personal communication, one of the main authors of the handbook has confirmed that they are not aware of any source that could lead to an estimation of these figures.

Besides the information provided in the AWARENET Handbook, some sporadic information has been encountered at the country level - we give here a brief overview of this information.

In Austria, data are available on the generation of the following waste streams:

- Kitchen and canteen waste
- Food, beverage and tobacco waste
- Waste from vegetable and animal fat products
- Waste from anima husbandry and slaughter
- Skins and leather waste
- Other waste from the processing and refinement of animal and vegetable products

However, no information is provided on their disposal or recovery.

In the Belgian region of Flanders, data is provided on industrial waste of animal or vegetable origin, split up according to the sector of origin and its destination. However, there is no matrix linking the origin of waste to its destination. There is no systematic information on kitchen waste from restaurants. A detailed study has been undertaken in 2005 on the market for animal by-products in Flanders, but it is not clear which fraction of these by-products falls within the scope of the definition of bio-waste. There is also fairly good information on the management of fried fats and oils in Belgium.

No quantitative data is available in industrial food waste in France, but the waste management industry has pointed out that most of this waste is recycled in technical applications.

Ireland provides data on commercial organic waste, but not on waste from the food industry.

Latvia has some data on the landfill of waste from food preparation and products.

In Lithuania, official data exist on the generation of waste from

- Meat and fish processing
- Waste from the sugar industry
- Waste from the alcohol industry

An independent study (Juškaitė-Norbutienė et al. (2007)) has provided detailed estimates on bio-degradable waste and by-products from food industry management systems, and on their treatment. The scope of their study is limited to Lithuania. To the best of our knowledge, this is the only comprehensive study on this topic that covers an entire country.

As pointed out in Section 4.7, Poland publishes data on the recovery and disposal of waste from, *inter alia*, food preparation and processing. However, this data is not split according to the sector of origin.

Finally, in the UK, detailed information is provided on the different disposal and recovery rates for commercial and industrial uses.

Thus, in total, just 6 MS provide national data that fall within the scope of “waste from the food and catering industry”. Data on the relative importance of different recovery or disposal techniques are almost completely lacking. When they do exist, they refer to a specific point in time – no time series are available.

5 The baseline

5.1 Goal

The goal of this assessment is to achieve data on the presumed evolution both in the generation of bio-waste, its collection and its treatment for all 27 Member States. As a start, data from 2006 are considered where available. This is the most recent year for which sufficient data is available in most of the Member States. Trends are calculated until 2020.

As requested in the Terms of Reference to this study, the baseline scenario is developed on the assumption that all Member States are coping with the targets from the Landfill Directive and with the policy targets they have imposed on themselves. This will request for many Member States a considerable and persistent effort on developing alternatives for landfill, on top of what is already undertaken. The baseline scenario should therefore not be considered as a business-as-usual scenario, as, there is a risk that business-as-usual may not lead to compliance with the Landfill Directive in some Member States

It has been argued by some that a Biowaste Directive could be necessary as a tool to support Member States in reaching the targets. This is a political question that we do not consider.

The impact assessment that is developed as a comparison of the baseline scenario with other possible scenarios, is an exercise to see what the costs and the benefits of the different scenarios will be, assuming that in the Baseline, the Landfill Directive targets are reached. As such, under anticipated Baseline scenarios for separate collection of biowaste, the Landfill Directive targets imply a given minimum quantity of treatment of biowaste as residual waste through means other than landfill. Whether these targets can be reached in the absence of a Biowaste Directive falls outside the scope of this study.

With the above, the impacts and potential benefits of a Biowaste Directive may be anticipated to exceed those as shown within this report

5.2 Overview

Trends in the generation, collection and treatment of bio-waste depend on following elements:

- **Generation of household or municipal waste**: This can be assessed based on
  - The actual generation of this waste fraction in 2006
  - The demographic evolution
  - The evolution in economic welfare, as an indicator for changing consumption patterns. The GDP is used as an indicator.
  - The degree of coupling or decoupling from both driving forces. No decoupling from demographic evolution is presumed, decoupling from economic evolution is presumed in some scenarios
- **Degree of coverage of household waste collection**. Although full coverage of all households is aimed at, in certain Member States this coverage has not yet been reached, and waste is treated by uncontrolled landfilling, incineration or reuse by the

producers. The developed scenarios only focus on waste entering the regular waste collection and treatment sector, excluding illegal treatment.

- **Composition of mixed household waste and its content of bio-waste**, and the fraction of bio-waste that is collected separately.
- The amount of **bio-waste generated by food industry and by catering activities**
- The actual and future distribution of **bio-waste treatment** over
  - o Landfilling (taking into consideration compliance with the Landfill Directive)
  - o Incineration
  - o MBT with aerobic or anaerobic biological treatment, mostly followed by disposal of the digestate
  - o Composting, or the aerobic generation of compost out of separate bio-waste fractions
  - o Anaerobic direction or the anaerobic generation of methane and digestate out of separate bio-waste fractions, often in co-treatment
  - o Home composting.

Specific treatment methods for bio-waste only apply to the fraction of separately collected waste. For the fraction of bio-waste included in the mixed household waste fraction, only general treatment methods like landfilling, incineration or MBT[32] (which essentially is a preparatory activity before landfilling or incineration) can be considered.

5.3 Separate steps in developing the baseline scenarios

The definition of the baseline scenario for each of the Member States passes through following stages:

- Inventory of available basic quantitative data, and assessment of lacking information.
- Classification of the Member State on its stage in a typical waste policy development, e.g. its position on the Kuznetz curve, its degree of decoupling.
- Assessment of the presumed generation of bio-waste between 2006 and 2020.
- Assessment of changes in waste collection and waste treatment based on available policy information.
- Assessment of bio-waste treatment capacities needed in future.

5.4 Detailed modelling

5.4.1 Assessment of the evolution of the bio-waste generation

To assess the future evolution in waste generation, the assumption of a linear relationship between population and total waste production is made. We start from the presumption that no decoupling takes place between demography and waste generation. The total waste production W_{total} depends on the demographical evolution P in the year 20xx, and a linear correspondence between both parameters is assumed. The total waste production can be expressed as a function of the average waste production $W_{average}$ per capita and the total population number P.

Equation 1: Waste production and population

[32] Composting of mixed household waste, which is applied in some regions in France and Spain, and which is orientated towards generation of compost in stead of stabilising waste, is considered as a form of MBT, because it corresponds technically with MBT technology.

$$W_{total}^{y=20xx} = W_{average} * P^{y=20xx}$$

The total waste generation figure needs to be corrected for the collection coverage of the population, as we do not consider illegal or wild waste treatment. For each Member State a 100% coverage is assumed, or a progressive evolution towards a 100% coverage. The considered total is the generated W_{total} multiplied with the degree of collection coverage C in the considered year 20xx. C has a value between 0 (no coverage) and 1 (100% coverage)

Equation 2: Covered and total waste

$$W_{total\,covered}^{y=20xx} = W_{total}^{y=20xx} * C^{y=20xx}$$

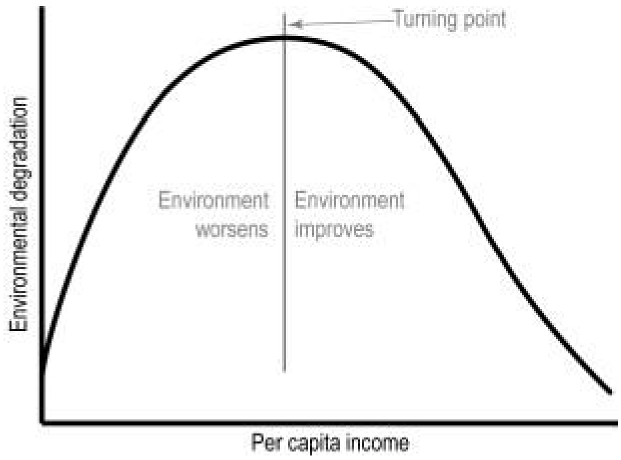
5.4.2 Assessment of the relation between waste generation and GDP

The evolution of the average amount of waste generated is defined by the economic evolution. It is expected that the amount of waste generated is related to the economic position of the consumer. Therefore the average waste generation per capita does not remain stable over the tested period, but evolves in line with growing welfare.

The average waste production per capita will, in function of the economic development, evolve towards the average total waste production as it is observed nowadays in the economic front runner Member States in Western European countries. The Flemish figure for average waste production per capita could be taken as an indicator: in 2007 this is 555 kg/inh.year.

Some Member States have a similar waste generation, stabilised at a comparable level. In other Member States a growth towards this stabilised maximum can be expected. This growth will not be linear, but will be more quickly in the beginning and will slow down at the end. As observed in Flanders, due to technical improvements, public awareness and growing prevention of waste, the average municipal waste generation per capita will ceil at a certain value.

This evolution is described in literature as the first half of a so-called environmental Kuznetz curve. We do not make the supposition that after reaching a stable top the waste generation will spontaneously diminish, because this is not observed empirically in the western European countries.



Source: www.maf.govt.nz

Figure 5-1: Traditional Kuznetz curve

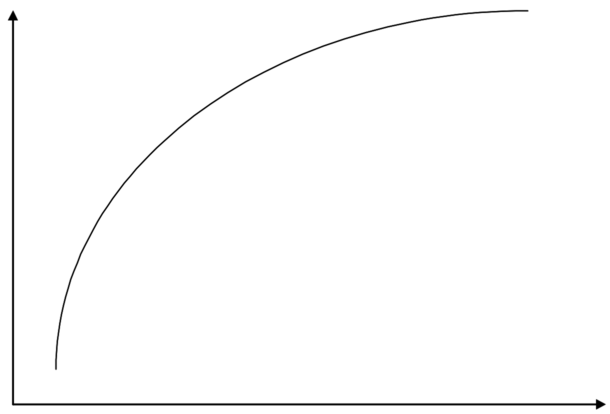


Figure 5-2: Curve for growth of average waste production to a stabilised maximum

To describe the dependency of the waste growth on the gross domestic product different scenarios can be described. The scenarios are differentiated according to the decoupling between the waste generation and the GDP as an indicator for economic activity. The concept of decoupling, as defined by the OECD, distinguishes between:

- Negative decoupling: the waste production grows faster than the economy
- No decoupling: the waste production and the economy grow at the same speed
- Relative decoupling: the waste production grows more slowly than the economy
- Absolute decoupling: while the economy is growing, the waste production is diminishing.
- Sustainability: the waste production has diminished towards a minimum level where nature's absorption capacities are not endangered in future.

We divide Member States, based upon the collected qualitative and quantitative information, into four different classes upon which different scenarios can be applied. In these scenarios we keep up with a long term perspective and do not take into account the financial crisis and short term disturbances in the economic evolution (see Section 4.2).

- A scenario 1 where in a first phase, due to quick economic growth and a catch up operation in a context with less environmental awareness or pressure, a negative decoupling takes place and waste generation grows more quickly than the economy. This first phase is followed by stabilisation.
- A scenario 2 no decoupling takes place and the environmental impact evolves at the same speed as economic activity.
- A scenario 3 occurs where the waste generation is decoupled from the economic growth (relative decoupling) and tends to stabilise around a maximum value. The only factor influencing the waste quantity is the demographic growth.
- A scenario 4, tending to sustainability, occurs where the waste generation diminishes before it stabilises at a certain value, while the economy grows (absolute decoupling)

These four scenarios can be translated into a mathematical model. As scenario 4 has not been used in this study, we will not discuss it further.

Scenario 1

In scenario 1 the growth rate of the GDP is applied to the actual average waste generation (A). Using a linear extrapolation the moment is discovered when the waste production would equal the actual waste production in Flanders (B), which we took as a benchmark figure. Thus a time frame for the waste production evolution can be assessed.

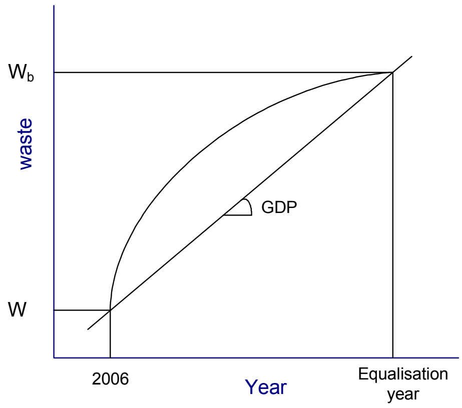


Figure 5-3: Assessment of equalisation year and prospected evolution in scenario 1

The evolution for the growth of the average waste production under scenario 1, with negative decoupling in the first phase and stabilising at a maximum in the equation year, can be calculated starting from the actual average production, the benchmark value, and the growth of the GDP.

The evolution as described in Figure 5-3 is approached as in Table 5-1.

Table 5-1: Growth of waste production under scenario 1

First fifth of period	Growth of waste production = GDP growth * 2
Second fifth of period	Growth of waste production = GDP growth * 3/2
Third fifth of period	Growth of waste production = GDP growth
Fourth fifth of period	Growth of waste production = GDP growth * 1/2
Fifth fifth of period	Growth of waste production = 0

Scenario 2

In this scenario waste grows at the same speed as economic activity, only to stabilise when the equation year is reached. In this scenario, we assume no decoupling over the whole period between 2006 and the equalisation year, including a stabilisation period at the end in which relative decoupling is achieved. This would entail a slight negative decoupling in the first phases.

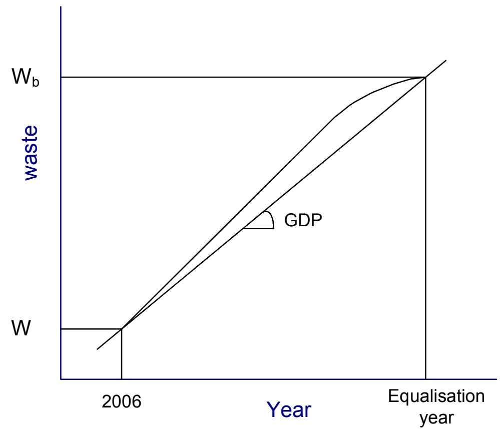


Figure 5-4: Assessment of equalisation year and prospected evolution in scenario 2

The growth percentage of scenario 2 is altered to incorporate the effects of a first period of quasi no decoupling and a final stage of stabilising at a maximum.

Table 5-2: Growth of waste production under scenario 2

First fifth of period	Growth of waste production = GDP growth * 1,3
Second fifth of period	Growth of waste production = GDP growth * 1,3
Third fifth of period	Growth of waste production = GDP growth * 1,3
Fourth fifth of period	Growth of waste production = GDP growth * 0,85
Fifth fifth of period	Growth of waste production = GDP growth * 0,3

Scenario 3

In scenario 3 the average waste generation approaches the benchmark value, which means that 2006 and the equalisation year are close. The average waste generation has reached a maximum value and is rather stabilised around it. Decoupling has occurred because the GDP continues growing while the average waste generation does not grow, disregarding certain fluctuations.

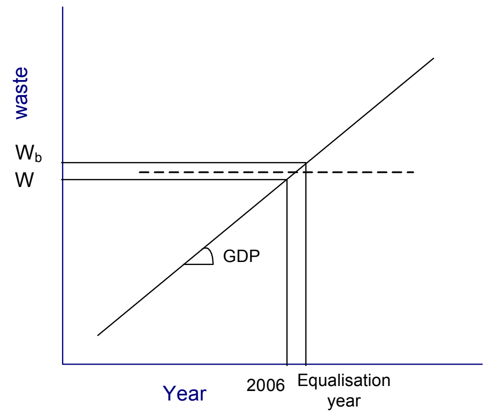


Figure 5-5. Assessment of equalisation year and prospected evolution in scenario 3

Table 5-3: Growth of waste production under scenario 3

First fifth of period	Growth of waste production = 0
Second fifth of period	Growth of waste production = 0
Third fifth of period	Growth of waste production = 0
Fourth fifth of period	Growth of waste production = 0
Fifth fifth of period	Growth of waste production = 0

The assessment of the total waste production can be made by multiplying the average production to the total population number for each year, and correcting for the collection coverage.

The total amount of municipal bio-waste generated is assessed as the total covered generation of municipal waste multiplied with the assessed ratio total bio-waste/total municipal-waste. This ratio is obtained from the evolution of the composition of the total waste generation.

5.4.3 Assessment of the evolution of bio-waste treatment

Based on the quantitative information on available and planned capacities and the qualitative information of policy goals and targets, the expected distribution of bio-waste treatment techniques needs to be drafted for each Member State.

In this exercise we take into account:

- The intended compliance with the Landfill Directive
- The policy preferences of the Member States

5.4.3.1 Compliance with the Landfill Directive

We presume the intended compliance with the provisions of the Landfill Directive, on the amount of bio-waste that may be landfilled. Article 5 point 2 of the Directive states target values for the reduction of the amount of bio-waste landfilled, referring to the values generated in 1995.

Table 5-4: Limit years to reach the reduction of landfilling biodegradable waste imposed by the Landfill Directive

MS	Max 75% of generation in 1995 landfilled.	Max 50% of generation in 1995 landfilled.	Max 35% of generation in 1995 landfilled.
AT	2006	2009	2016
BE	2006	2009	2016
BG	2010	2013	2020
CY	2010	2013	2020
CZ	2010	2013	2020
DE	2006	2009	2016
DK	2006	2009	2016
EE	2010	2013	2020
ES	2006	2009	2013
FR	2006	2009	2016

GR	2010	2013	2020
HU	2004	2009	2016
IE	2010	2013	2020
IT	2006	2009	2016
LT	2010	2013	2020
LV	2010	2013	2020
LX	2006	2009	2016
MT	2010	2013	2020
NL	2006	2009	2016
PL	2010	2013	2020
PT	2006	2009	2016
RO	2010	2013	2020
SF	2006	2009	2016
SK	2010	2013	2020
SL	2006	2009	2016
SV	2006	2009	2016
UK	2010	2013	2020

The targets of the Landfill Directive are set on biodegradable waste and not on bio-waste. This means that a reduction of 75% of biodegradable waste to be landfilled does not automatically request a reduction of 75% of biowaste to be landfilled. When e.g. the landfilling of paper waste is reduced with more than 75%, the targets of the directive can be reached when less than 75% reduction of landfilling biowaste is realised (see Section 2.1). In the modelling for the baseline scenario, we start however from the presumption that these effects are merely marginal, as we presume a ratio of biowaste/paper of more than 2 (see Section 5.4.3.3), and we presume that the recycling of paper and of biowaste will follow largely the same trends.

5.4.3.2 Policy Preferences

The preferences of the Member State for energy applications or compost applications are taken into account, as expressed in their national waste management plans. If no specific information was available, we have made reasoned assumptions, based upon the local needs for organic material or for energy on the one hand and extrapolations of the scenarios for similar countries on the other hand.

An evolution has been modelled from the actual situation towards the expected situation, both in quantities (based on the generation figures) and on shifting.

5.4.3.3 Evolution of the Composition of the Generated Household Waste

An important aspect, both for the assessment of the evolution in the quantity of bio-waste and in its possibilities for treatment, is the evolution of the composition of the generated household waste.

We take once again the composition of the waste in Flanders in 2006 as a benchmark. Household waste generation and composition are stabilised, and can give a good impression of a presumable evolution in some other Member States. Moreover reliable statistics are available to substantiate the presented figures.

The figures in Table 5-5 are based on:

- The relative importance of each fraction of waste that is separately collected in 2006, see column 2
- The results of a sorting exercise on mixed municipal waste in 2006, see column 3

- The weighed sum of both percentages, taking into account that 71.46% of the municipal waste is collected separately in 2006, and 28.54 is collected as mixed waste.

Table 5-5: Assessment of the fractions in generated municipal waste in 2006 in Flanders

selective collected fractions	selective collection %	mixed waste composition %	fractions in mixed waste	**total generation %**
glass	8,01	2,58	glass	**6,46**
paper cardboard	20,57	10,56	paper cardboard	**17,71**
plastics	2,46	13,88	plastics	**5,72**
vegetable, fruit and garden	12,97	39,43	bio waste	**35,90**
green waste	21,52			
beverage cartons	0,42	3,62	beverage cartons	**1,33**
textiles	1,53	4,51	textiles	**2,38**
dipers	0,10	9,03	dipers	**2,65**
construction and demolition	19,56	0	construction and demolition	**13,98**
small hazardous waste	0,73	0,55	small hazardous waste	**0,68**
metals	2,95	15,84	other	**13,19**
wood	5,48			
tyres	0,10			
sheet glass	0,39			
WEEE	1,97			
reusable waste	1,24			
	100,00	100,00		100,00
weighing		kg/inh	%	
mixed household waste 2006		153	28,54	
selective collected household waste 2006		383	71,46	
		536		

In Flanders 35.90% of the generated municipal waste is municipal bio-waste.

If we compare these figures with the composition of municipal waste in e.g. the region Pleven in Bulgaria in 2006, following result can be obtained:

Table 5-6: Composition of household waste in Pleven and Flanders

	Pleven	**Flanders**
Bio-waste	33,52	35,90
paper cardboard	9,93	17,71
Plastics	8,55	5,72
Glass	7,87	6,46
Metals	2,56	2,11
textile	3,64	2.38
inert	29,30	13,98
Other fractions	4,64	16,78
	100	100

For this Bulgarian case (Pleven can be considered representative for the whole of the country) this analysis would mean that the fraction bio-waste in the total generated municipal waste would evolve from 33.52% to 35.90%. The fraction can be considered as rather stable with a light tendency to augment.

The composition of the generated Flemish waste suggests roughly that bio-waste is 56% of the total bio-degradable waste (bio-waste+paper+textiles+1/2 other).

In what follows, we will always assume that bio-waste corresponds to 56% of total biodegradable municipal waste, unless there is country-specific information that points to the contrary.

5.4.3.4 Composition of the Generated Bio- Waste

For all countries, the same approach has been used to calculate the baseline shares of food and garden waste in each biological recycling option (centralised composting, AD and home composting):

- For each country, an estimate has been made of the share of garden and food waste in total biowaste generation (S_{GW} and S_{FW}).

- For each country, an assessment has been made of the selective capture rate of garden and food waste (SC_{GW} and SC_{FW}).
- The amounts of garden and food waste that are collected separately (C_{GW} and C_{FW}) have been calculated as (where BIO is total biowaste generated and COV is overall collection coverage):
  - C_{GW}= BIO * S_{GW} * COV * SC_{GW}
  - C_{FW}= BIO * S_{FW} * COV * SC_{FW}
- The amounts of food and garden waste that are treated in each biological recycling option has been calculated in such a way that:
  - The sum over all recycling options equals the amounts that are collected separately.
  - The shares of food waste in home composting and in AD reflect the characteristics of the mix and predominant nature of collection systems operated (and likely to be forthcoming) in individual countries.

5.5 Needed input information:

- An assessment of the average generation of household waste
- An assessment of expected demographic evolution
- An assessment of the average growth of GDP
- An assessment of the collection coverage in the different years
- An assessment of the ratio bio-waste/household waste in the different years
- An assessment of the policy views and the changes in treatment of bio-waste

5.6 Data Gathering Outcome

Based on available information in the public domain, a country fact sheet has been developed and a baseline scenario has been constructed, for all 27 Member States. The gathered basic information, the methodology and the outcome of this exercise were submitted for comments to the competent national authorities at the end of April 2009. The methodology was presented in a stakeholder conference on June 10. No changes or comments where received for Austria, Bulgaria, Cyprus, Estonia, France, Greece, Hungary, Ireland, Italy, Latvia, Luxembourg, Malta, Poland and Slovakia. The baseline scenarios were revised for the three Belgian regions, Czech Republic, Denmark, Germany, Finland, Latvia, Portugal, Romania Slovenia, Spain, Sweden, the Netherlands and the UK.

5.7 Summary of the Results

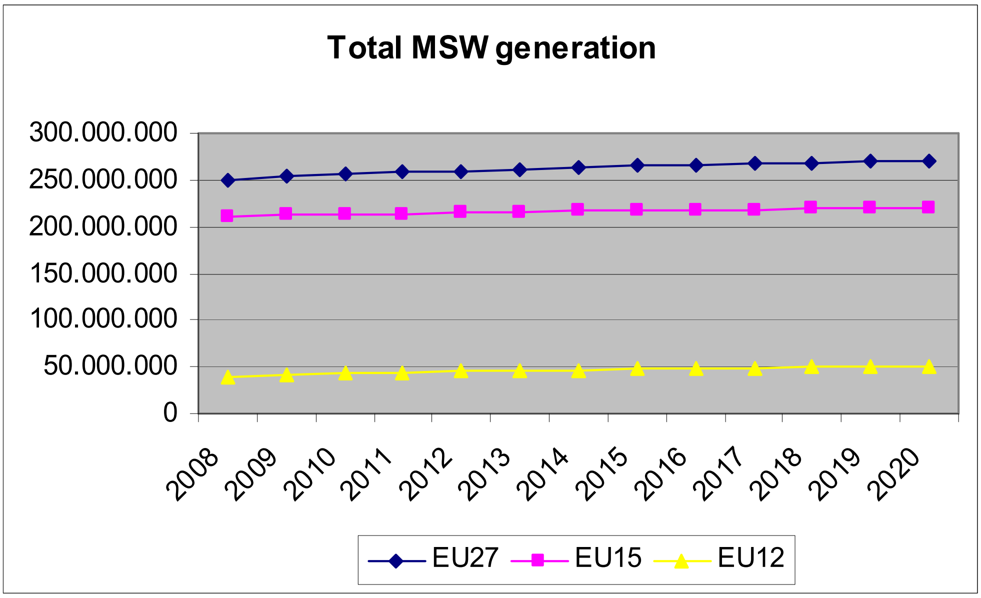
We refer to Annex A of this report for a detailed discussion of the baseline for each individual Member State.

Here, we limit ourselves to the presentation of some summary tables and graphs.

Table 5-7: MSW generation in the baseline (in tonnes)

	EU27			EU15			EU12		
	Biowaste	Non-biowaste	Total MSW	Biowaste	Non-biowaste	Total MSW	Biowaste	Non-biowaste	Total MSW
2008	87.718.367	162.509.636	250.228.003	74.825.380	136.300.793	211.126.173	12.892.987	26.208.843	39.101.830
2009	89.020.010	164.362.841	253.382.851	75.284.629	136.968.951	212.253.580	13.735.381	27.393.890	41.129.271
2010	90.026.473	165.913.194	255.939.667	75.669.620	137.632.920	213.302.540	14.356.853	28.280.274	42.637.127
2011	90.914.846	167.127.565	258.042.411	76.021.474	138.201.469	214.222.943	14.893.372	28.926.096	43.819.468
2012	91.801.186	168.322.808	260.123.994	76.374.443	138.771.359	215.145.802	15.426.743	29.551.449	44.978.192
2013	92.549.266	169.333.957	261.883.223	76.668.574	139.250.904	215.919.478	15.880.692	30.083.053	45.963.745
2014	93.219.592	170.241.734	263.461.326	76.963.568	139.731.121	216.694.689	16.256.024	30.510.613	46.766.637
2015	93.847.413	171.036.366	264.883.779	77.230.342	140.119.649	217.349.991	16.617.071	30.916.717	47.533.788
2016	94.418.616	171.756.911	266.175.527	77.497.766	140.508.177	218.005.943	16.920.850	31.248.734	48.169.584
2017	94.988.912	172.459.206	267.448.118	77.765.836	140.896.705	218.662.541	17.223.076	31.562.501	48.785.577
2018	95.558.307	173.159.548	268.717.855	78.034.560	141.285.233	219.319.793	17.523.747	31.874.315	49.398.062
2019	96.070.837	173.785.870	269.856.707	78.303.930	141.673.762	219.977.692	17.766.907	32.112.108	49.879.015
2020	96.582.909	174.410.947	270.993.856	78.573.953	142.062.290	220.636.243	18.008.956	32.348.657	50.357.613

Graph 1: Total MSW generation in the baseline



Graph 2: Total biowaste generation in the baseline

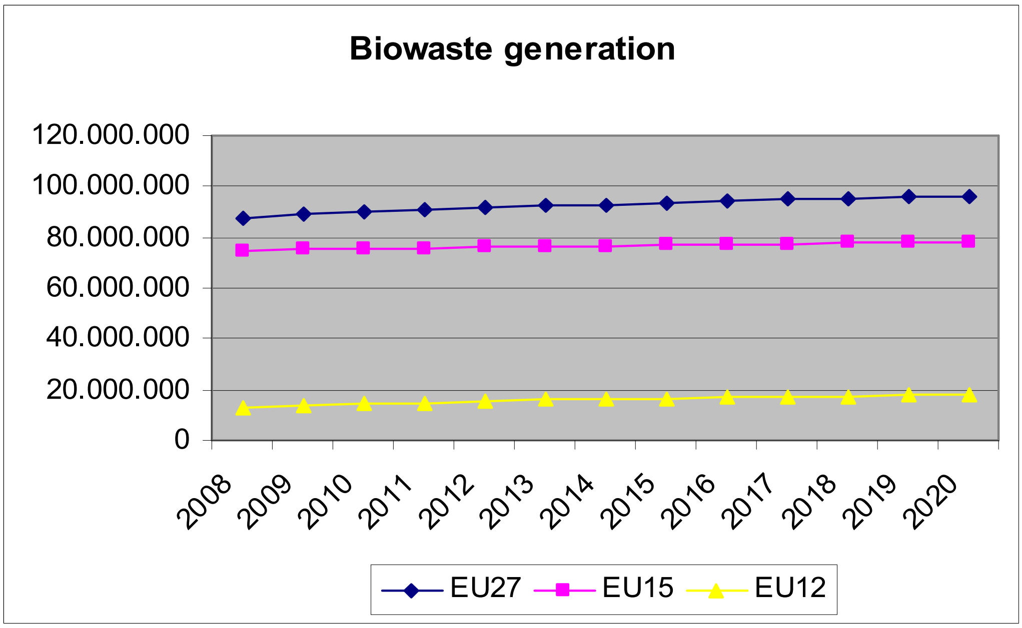


Table 5-7 indicates a steady increase in MSW and in biowaste generation, both in the EU15 and EU12. Between 2008 and 2020, biowaste generated increases with 10% at the EU27 level, with 5% at the EU15 level, and with 40% at the EU12 level. However, the share of the EU12 in total biowaste generated remains below 19%. According to Eurostat, the population of the EU12 is approximately 21% of the population of the EU27- this suggests that by 2020, the differences in income per capita will not be the main driver of the differences in waste generation.

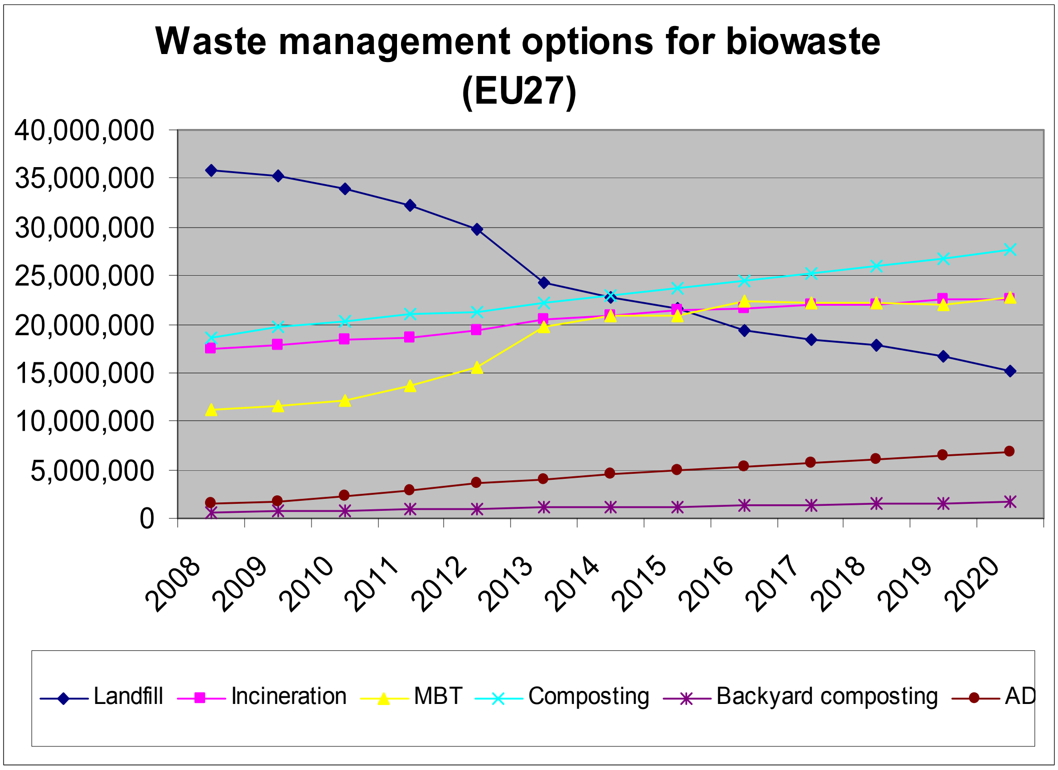
Table 5-8: Waste management options in the baseline

	Landfill	**Incineration**	**MBT**	**Composting**	**Home composting**	**AD**
2008	35.738.844	17.402.551	11.211.481	18.658.979	654.508	1.534.937
2009	35.171.916	17.728.458	11.473.431	19.642.019	724.430	1.722.447

2010	33.998.654	18.377.499	12.066.505	20.266.791	797.920	2.258.470
2011	32.240.606	18.619.624	13.697.095	20.949.277	883.454	2.767.027
2012	29.689.359	19.420.645	15.493.077	21.298.551	966.487	3.654.014
2013	24.346.536	20.513.166	19.652.181	22.123.592	1.046.651	4.073.340
2014	22.832.121	20.764.563	20.778.080	22.908.972	1.120.475	4.457.254
2015	21.635.520	21.401.242	20.936.875	23.674.209	1.197.558	4.857.487
2016	19.247.054	21.631.446	22.444.891	24.430.271	1.277.907	5.272.295
2017	18.440.031	22.029.764	22.198.214	25.168.652	1.355.195	5.703.790
2018	17.837.002	22.032.829	22.113.687	25.962.491	1.439.964	6.103.070
2019	16.651.462	22.584.068	21.989.103	26.748.914	1.530.648	6.520.225
2020	15.121.568	22.552.781	22.771.721	27.600.154	1.626.861	6.885.235

* Differences in the totals between Table 5-7 and Table 5-8 are due to households not connected to collection services.

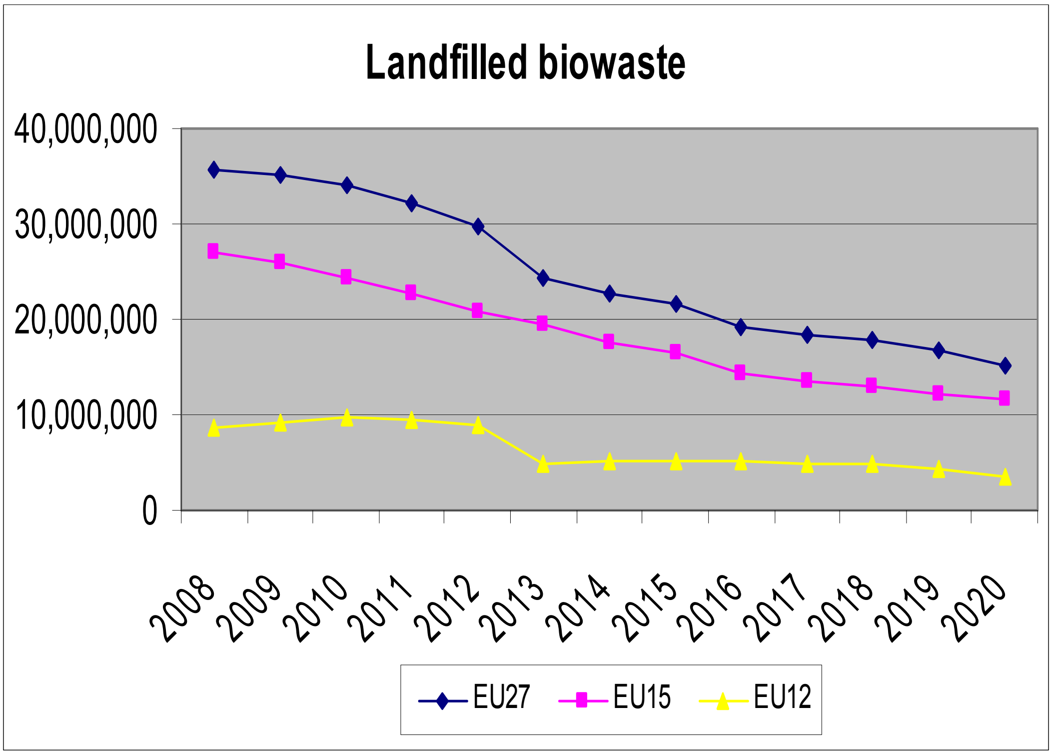
Graph 3: Waste management options in the baseline (EU27)



* Differences in the totals between Table 5-7 and Table 5-8 are due to households not connected to collection services.

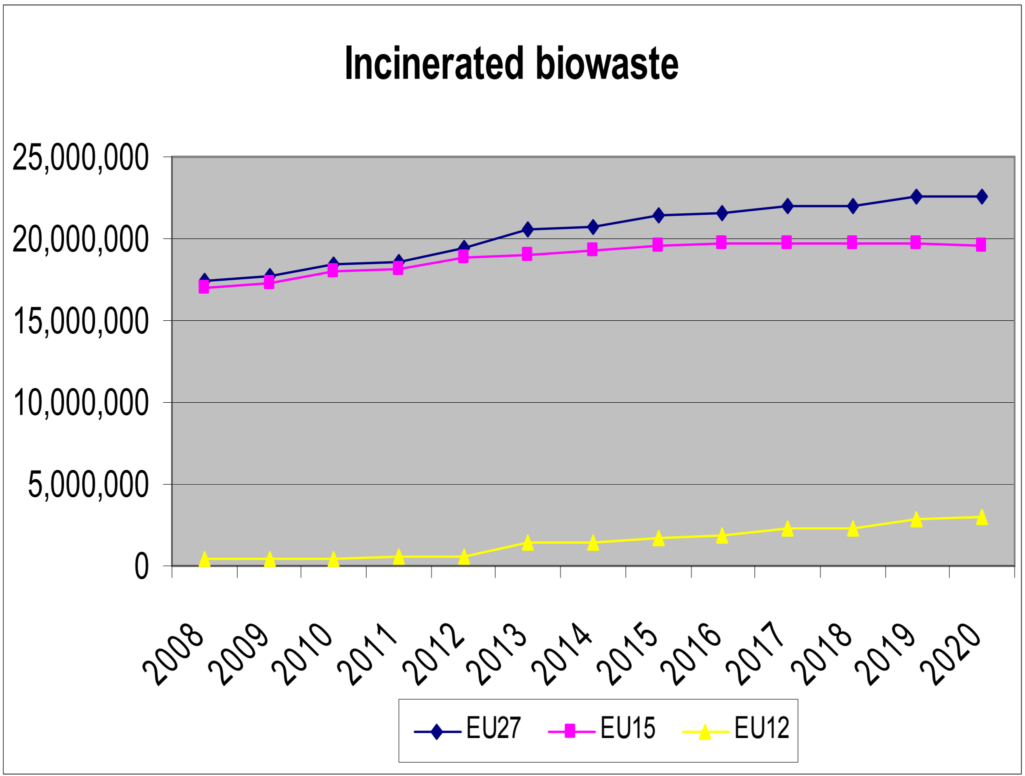
Graph 3 shows that the amounts going to landfill will steadily decrease through time (-40%). It is clear that the Landfill Directive is an important driver, although our discussion of individual countries shows that several Member States go beyond these targets. The most significant increases in absolute value are noted for composting (plus 11 million tonnes) and MBT (plus 7 million tonnes). AD and home composting increase by more than 200% and 300%, but remains small in relative terms. Incineration remains relatively stable (+12%).

Graph 4: Landfilling in the baseline



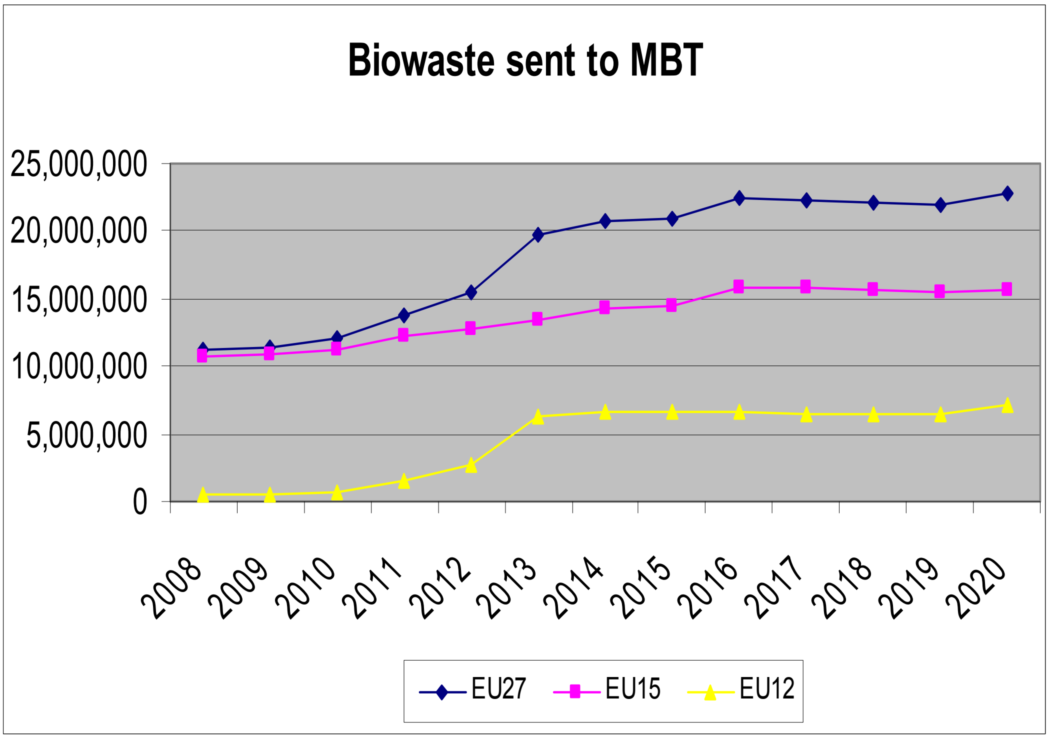
In Graph 4, we see that the drop in landfilling is the most pronounced in the EU15 in absolute terms, which reflects mainly the EU15's economic and demographic weight. Indeed, in the EU12, landfilling decreases by almost 40%, which is an important relative change.

Graph 5: Incineration in the baseline



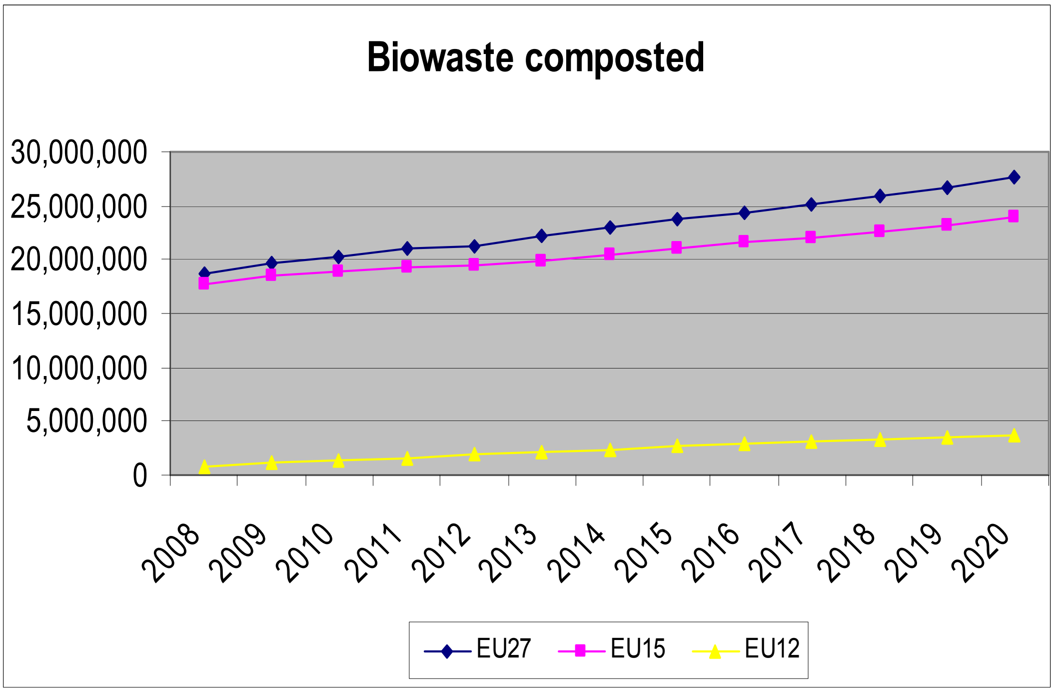
Even by 2020, the bulk of incineration (86%) takes place in the EU15. The share is thus much higher than what could be expected on the basis of demography alone. The high capital intensity of incinerators and the high existing capacities in the EU15 can help understand this high share.

Graph 6: MBT in the baseline



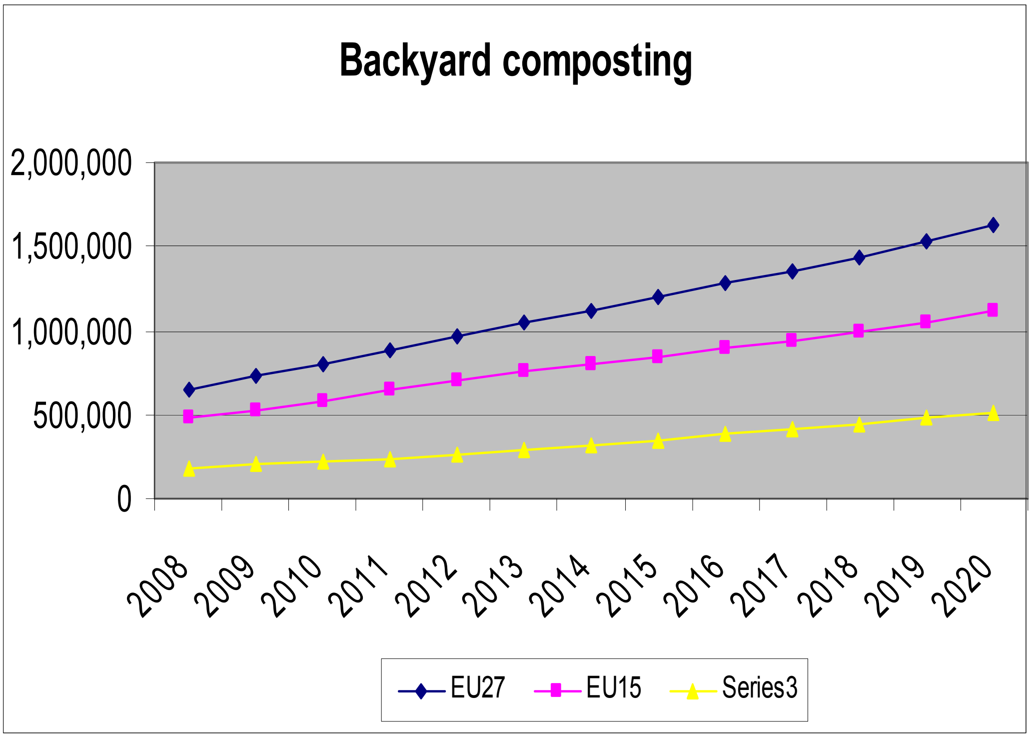
MBT increases significantly, both in the EU15 (by 45%) and certainly in the EU12 (almost 15 fold).

Graph 7: Composting in the baseline



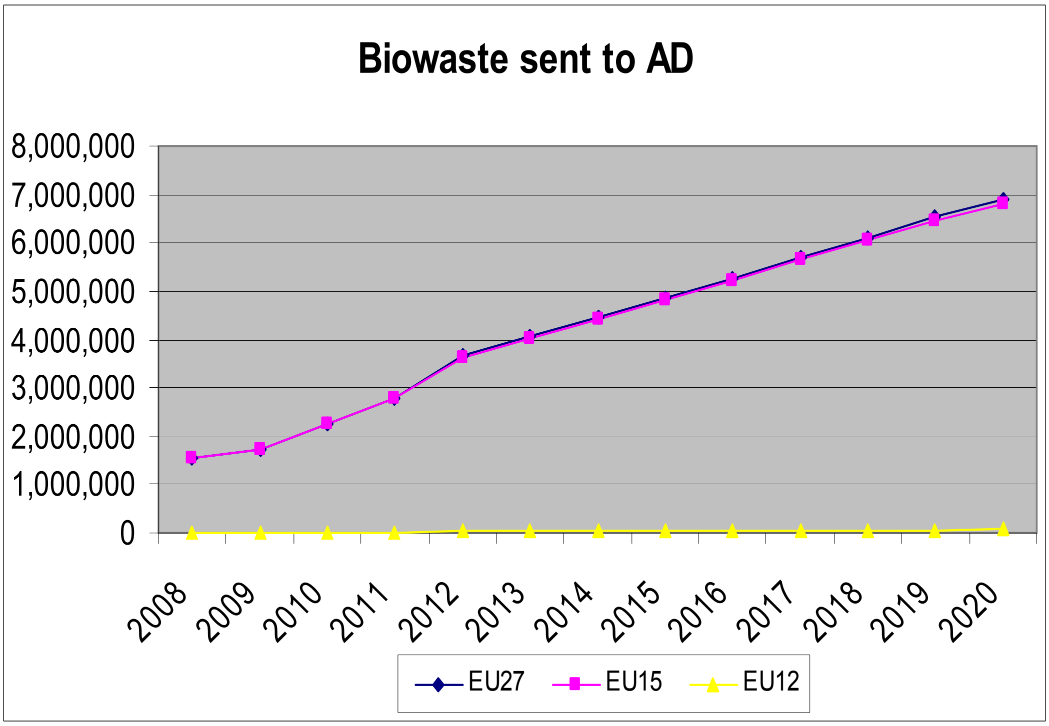
Composting increases by 34% in the EU15 and 4 fold in the EU12.

Graph 8: Home composting in the baseline



Home composting increases more than two fold in the EU15 and almost three fold in the EU12. In absolute terms, the quantities composted in the backyard in the EU12 almost reaches the quantities in the EU15 by 2020.

Graph 9: Total AD in the baseline



AD increases 4 fold in the EU15, and grows to 66 Ktonnes from virtually zero in the EU12. However, the quantities of biowaste sent to AD in the EU12 remain extremely small compared to quantities sent in the EU15.

5.8 Assessment of the baseline scenario

5.8.1 Uncertainty with respect to the current situation

As illustrated at length in Section 4, data on the subject at Member State level is sometimes sketchy, not always accurate, and definitions and reporting methods are not harmonised across MS. This means that a wide range of uncertainty surrounds the starting point of the baseline construction. Provisions for a more clear definition and for data gathering and reporting using the definition can contribute in solving the problem.

In the case of waste from industry, the project team has estimated that constructing a baseline from the information that is available in the public domain would be too arbitrary to have any credibility. We have therefore proposed to move to a case study approach, using well checked figures, instead of a EU27 wide forecast.

Independently of the previous observation, one important policy conclusion is that the **existing harmonised reporting requirements are not adapted to the data needs of this type of study**. This is partially due to the fact that the subject of this study (bio-waste) is more restrictive than the concept of biodegradable waste used in the Landfill Directive, and is not in line with the definitions as applied in the Waste Statistics Regulation.

5.8.2 Uncertainty with respect to policy intentions

A lot of uncertainty also surrounds the policy intentions of the MS. This is partially due to the fact that several MS do not announce their policy intentions in the public domain. However, we have also come across several MS where policy makers have not yet made up their mind on the way forward for their country. This is particularly the case with countries that have a history of heavy reliance on landfilling, and which have to decide in the near future what fundamental options to take.

Some Member States refer to the need for a EU-wide legal driver, giving the necessary certainty and legal security for investments and drawing up a policy on a long term basis.

In this respect, it is not clear to us how we should interpret the relative low response rate to the first draft of the baseline (see Section 5.6).

5.8.3 Wide variety of policy approaches

We have observed a wide variety of policy approaches to biowaste. Section 5.7 has shown that, between the old and the new MS, we can expect these differences to remain significant in the future.

Differences in policy approaches can be due to several, not mutually exclusive, factors.

First, preferences may vary across countries. Each approach has specific benefits and costs. Even if private benefits and costs would be uniform across countries, policy makers may have different views on the trade-offs between these costs:

- More affluent and densely populated areas may attach relatively more importance to disamenity effects than to the capital investments that are needed to mitigate these effects.
- Depending on the sanitary risks they are willing to take, MS may have a more or less restrictive interpretation of the ABPR. This affects the treatment methods that will be

chosen, and also whether kitchen waste will be included in the separately collected fractions.

Second, some selected policy options depend upon locally defined benefits and costs:

- All other things being equal, countries with important wind power, solar power hydropower or biomass resources and potential will attach relatively less importance to the recovery of energy from waste management. This will be discussed at length in Chapter 7.
- The alternatives to energy recovery from waste management will also affect the environmental benefits of energy recovery – these are clearly different when the energy source displaced is coal[33] than when it is natural gas.
- Often, costs for treatment are sunk, this means that they cannot be recovered if one would move waste away from this treatment option. This means that waste authorities that have to choose between (for instance) investing in incineration or in composting capacity face a completely different trade-off than countries that have already invested in incineration capacity: within the economic lifetime of an incinerator, the capital investment cannot be recouped. Therefore, it is possible that a waste authority with relatively large incineration capacities reaches the conclusion that incineration is the preferable option for the future as well. However, this does not imply that incineration should also be the preferred option for an otherwise identical country that has not yet invested in sunk capacities. Moreover, the “sunkness” of an investment depends on the time perspective one takes, and on the level of geographical aggregation. This will be discussed at length in Section 7.11.4.
- The benefits of compost vary across MS. This can be due to differences in climatic conditions or in soil quality – in countries that are not affected (yet) by droughts or erosion, the benefits of compost application are less obvious than in others. These benefits also depend on the competition from other fertilisers (such as manure) and on limitations on the amounts of nitrogen and phosphorus that can be applied to the land. As the market value of compost is low compared to transportation costs, this implies that local factors will play an important role in the preferences for compost.
- The existence of AD capacity also depends on the quantities of excess manure supply that need to be disposed off. However, manure falls outside the scope of the definition of bio-waste.

5.8.4 The interaction with the Landfill Directive

At first sight, one would expect the Landfill Directive to have a positive effect on the recycling of biowaste.

However, the situation is more complex:

- The targets refer to one single reference year (1995), for which reliable estimates of BMW do not always exist - MS have therefore an incentive to exaggerate their estimates for 1995. For some countries, independent experts have indicated that this is indeed what has happened.
- The Landfill Directive refers to biodegradable waste. Therefore, in order to reach their targets, MS will concentrate on the waste streams that are the easiest to divert from landfill, and will not undertake a comprehensive cost-benefit analysis in order to determine priorities.

[33] Or oil shale as in Estonia.

- The targets refer to diversion away from landfill, and not to diversion to a particular treatment method. Therefore, MS have an incentive to choose the cheapest option, disregarding the environmental benefits and costs. This is more likely to be the case in countries with limited financial resources.

6 Definition of the policy scenarios

We give here an overview of the policy scenarios that were withheld for further analysis.

6.1 Principles to be respect in the definition of recycling targets

Our methodological proposal for recycling targets is based upon the principle that a target only makes sense if it can be measured; imposing targets requires subsequent monitoring and enforcement.

Existing statistics (including those from Eurostat) generally refer to the generation of biodegradable municipal waste, not to the generation of biowaste. Biodegradable municipal waste is both broader and narrower than bio-waste: on the one hand, biowaste includes waste from food processing plants (i.e. not municipal waste); on the other hand, biodegradable municipal waste also includes paper, cardboard and biodegradable textiles. For the purposes of this study, we have made reasoned assumptions with respect to the relation between biowaste and biodegradable waste (see Section 5.4.3.3). However, it is unlikely that such an approach would be accepted for a legally binding target. Therefore, if a recycling target is to be implemented, new reporting and monitoring requirements are required – applying a correction factor to biodegradable waste will not be enough.

For the **calculation of the quantities that are recycled**, the following fractions can be included:

- The inputs to compost production (minus rejects and recycling residues)
- The inputs to AD (minus rejects and recycling residues)
- The inputs to structured/supported home composting

Moreover, the compost should:

- Comply with a product standard
- Be used as a product

Only source separated waste is calculated into recycling targets.

6.1.1 The role of rejects

Rejects can be waste streams that are refused at the gate because they do not comply with the acceptance criteria in the environmental permit of the installations, or because they are technically not fit for an optimal composting operation. These waste streams can only be assessed through a careful registry of incoming and refused waste streams. Such a registry is not included, nor in the Waste Framework Directive neither in the IPPC Directive.

Recycling residues can be found at the outgoing side of the installation. Besides the recycled product 'compost' a more or less important side stream of non compostable waste can be generated that needs disposal or further treatment.

The exact rules for measuring rejects and recycling residues could be part of a Biowaste Directive or could be determined through Comitology procedure as a part of the legal implementation strategy. However, for the purposes of the simulations, assumptions need to be made with respect to the quantities of rejects and recycling residues.

For this stream we assume:[34]

- 10 % residues and rejects combined in the case of AD;
- 5% residues and rejects combined in the case of composting.

6.1.2 Compost standards

For modelling purposes, it must be assumed that one single compost standard applies (across member states for compost output). We will assume that meeting the standard implies source separation of biowaste.

Standards for digestate from AD are identical to the above mentioned standards for compost and stabilised biowaste, as they will as well cover nutrient content and hazardous substances. To meet standards, digestate will require post treatment stabilisation and also potential blending with green waste compost.

6.1.3 Home composting

In accordance with the terms laid out in the Waste Framework directive, we had initially proposed to consider home composting (since it is not collected as waste) as waste prevention and not as a recycling option.

However, studies are available to show that net welfare gains are to be made from home composting – see Section 7.3. Allowing home composting to be included in the recycling target accommodates the needs of areas with low population density.

However, if home composting is included as a recycling operation, we should define a way to measure it. Following problems are encountered:

- The figures for the actual generation of biowaste do not include waste treated through home composting. These figures are in most Member States based upon or derived from quantities of waste collected. Home composted waste is not collected and never enters an official waste treatment channel where it could be measured. Because we use the quantity of waste collected as a measure for the quantity of waste generated, we could for merely pragmatic reasons consider home composting as a measure of bio-waste prevention and therefore not as waste 'generated'.
- For some countries an important fraction of home composting could be assumed, where the fraction of bio-waste is low in the analysis of the composition of the municipal waste generated. It is of course unclear if this fraction that does not end up in the waste collection and treatment is really home composted (home composted) in a sound manner. This fraction could also be: given to home bread poultry, small livestock or other small scale private animal rearing; illegally disposed of through backyard incineration; dumped; disposed through sewer-based food-waste disposers
- In order to count home composting as an R3 recycling operation the quality of the compost should be ensured.

[34] Flanders applies a percentage for recycling residues for waste disposal taxes, which is 5% for composting of vegetable, garden and green waste, 5% for recycling food waste, 8% for AD. As it is a method to avoid illegitimate tax reductions the percentage is kept deliberately low to have a security margin

Home composting can only be included as a recycling operation under following conditions:

- It should be home composting under application of an official stimulation programme where home composting vessels are subsidised or distributed to the population, and with accompanying programmes for the right home composting techniques. It then could be counted or assessed based upon the number of vessels distributed and an average composting capacity generated through these vessels.
- Regular sample surveys and analyses should define which percentage of vessels is used and which percentage of home generated compost fulfils the quality requests.
- The total assessed amount of waste home composted should be added to the amount of municipal bio-waste generated. The percentage of home composting leading to compost fulfilling the standards should be added to the amount of bio waste composted.

During a coordination meeting between the project team and the Commission services on 05 June, the Commission services concluded that the technical discussion on the requirements for monitoring should be the subject of a Comitology procedure, maybe in discussion with Eurostat.

During this meeting, the Commission services also proposed to allow (not force) MS to include home composting in the measurement of the recycling target, because to do otherwise would discourage home composting.

Our modelling work therefore assumes home composting to contribute to the recycling targets.

6.1.4 Definition of the reference year

We defined the recycling target in relation to biowaste generated in a given year.

This means that we have excluded the following options:

- **A fixed reference year such as used in the Landfill Directive**. If the reference year lies before the recycling target was announced, no reliable data will exist for this reference year if the target is expressed with reference to a waste stream that is not routinely being monitored in all MS (such as is indeed the case of bio-waste). If the reference year lies after the announcement of the recycling target, there is a risk that policies and definitions will be adapted (maybe temporarily) in order to reduce the amounts to be recycled in later years.
- **A target where it is requested to always increase recycling compared to the previous year**. In such a system, the countries that already perform well will have difficulties reaching the targets. This also encourages slow and gradual improvement in environmental performance rather than maximising the potential benefits at earliest opportunities.
- **Using the (moving) average of waste generated in a fixed reference year and in a variable reference year**. This methodology would be unnecessarily complicated.

6.2 Common assumptions

Taking into account the principles discussed in Section 6.1, all policy scenarios that have been withheld for further analysis share a series of common assumptions.

- The compost that will be generated in this policy scenario will fulfil the same quality requirements as described in policy scenario 1 (see below)
- The amount of bio-waste *recycled* is assessed as the sum of
  - The input of bio-waste for compost production minus the rejects and recycling residues
  - The input to AD minus rejects and recycling residues, assuming that the digestate will be used as a recycled product or that it will be further composted and used as compost.
  - The input of bio-waste for home composting
- Bio-waste can be considered as recycled under the condition that the compost or digestate generated at the end of the process will effectively be used or marketed as a product.
- Only waste that is collected source separated can be recycled.
- Use of digestate or compost that does not fulfil the quality standards for free use, and that is applied as an intermediary or final landfill cover, is not considered recycling.
- The output of the MBT process can no longer be considered as bio-waste and therefore the recycling of MBT-output is not included in the scope of this study on recycling bio-waste.
- Home composting is a recycling method which does not require collection. It is considered recycling under the conditions that the home composting programme is well funded, well structured and monitored. Allowing home composting to be included in the recycling target accommodates the needs of areas with low population density.
- We assume that the quantities of food and garden waste that go to every recycling option never drop below the quantities assumed in the baselines (except due to waste prevention). In other words, it is assumed that no switch from AD to composting (or vice versa) takes place: all changes in the quantities are due to increases in selective collection.
- Following timeframe is set for the recycling targets and for prevention.
  - No deviation from the baseline is expected as long as the recycling targets do not enter into force.
  - As starting date for the deviation from the baseline policy scenarios, we will take 2013. 2017 would then be the date for the interim targets and 2020 the date for the final target.
  - We assume that four years after the entry into force, an interim target has to be met which corresponds to 40% of the distance between the start value in 2013 and the final target.
  - We assume that progress between the different targets will be piecemeal linear.

The practical implementation of these assumptions is discussed more in detail for each scenario.

6.3 Scenario 1: Compost standards

Compost standards represent a valuable quality framework within which other policy measures can more successfully be enacted. As such, it will be assumed that any additional option under analysis is combined with a compost standard. However, we think it also important to assess compost standards as "stand alone" instrument, in order to have a better understanding of the value added of the other instruments under consideration.
This assessment is undertaken in Chapter 8.

6.4 Scenario 2 "High Prevention and Recycling"

This scenario includes:

- A reduction of biowaste generation to be treated with 7.5% compared to the baseline scenario; it will be assumed that this reduction is the result of prevention actions whose financial costs will not be included quantitatively in the model (though the environmental benefits are modelled according to Section 7.2.3).
- Home composting will contribute to the recycling target and remains as in the baseline.
- Separate collection of 60% kitchen waste and 90% of green waste. We will take into account that the recycling residue of bio-waste is no longer a bio-waste.
- We will assume gradual implementation of the targets for prevention and separate collection, with 1 January 2013 as start date. 1 January 2017 will be taken as the interim date. The interim target corresponds to 40% of the final target.
- It is assumed that all garden waste that is collected separately *in addition* to the baseline is recycled in IVC.
- It is assumed that all food waste that is collected separately *in addition* to the baseline is sent either to IVC, or to AD, depending on the recycling option that yields the highest net benefits to society. "Net" benefits include financial costs, but also environmental costs, including those related to the emissions of greenhouse gasses (GHG).

Scenario 2a is identical to scenario 2, except that all food waste that is collected separately in addition to the baseline is sent to AD, which is the recycling technology that yields the highest benefits in terms of GHG emission reductions. Under scenario 2a, there is thus a much higher focus on the potential of biowaste recycling as a tool to reduce GHG emissions.

6.5 Scenario 3 "Low Recycling"

Scenario 3 and 3a follow the same general approach as set out for scenario 2, with two differences:

- For the definition of the "low" recycling target for 2020, we use the midpoint between the current EU27 average and the current rate in the member state with the highest recycling rate (36.5%). All countries below currently the European target of 36.5% biowaste collection are assumed to reach this target by 2020. This is achieved by increasing garden waste collections up to the maximum of either the baseline situation or 70% of garden arisings, and any further diversion required is assumed to occur through food waste collection. Countries above the 36.5% target continue as per the baseline collection

- Prevention will not be considered.

In scenario 3, it is assumed that all food waste that is collected separately *in addition* to the baseline is sent either to IVC, or to AD, depending on the recycling option that yields the highest net benefits to society. “Net” benefits include financial costs, but also environmental costs, including those related to the emissions of greenhouse gasses (GHG).

Scenario 2a is identical to scenario 2, except that all food waste that is collected separately in addition to the baseline is sent to AD, which is the recycling technology that yields the highest benefits in terms of GHG emission reductions. Under scenario 2a, there is thus a much higher focus on the potential of biowaste recycling as a tool to reduce GHG emissions.

Thus, compared to scenarios 2 and 2a, this scenario is much less ambitious.

6.6 Industrial waste

Bio-waste from industrial origin can be divided into several main fractions.

- Waste comparable with the waste generated by common households: food and kitchen waste from canteens, garden waste from maintenance of the industrial premises…
- Waste from industrial sectors comparable to the working of a common household: catering, restaurants, maintenance of green spaces…
- Specific pre consumer waste streams from food industry, sludges, harvest remains, beet pulp, malt mash, processed food, off spec production, from the production of biodegradable garden and park waste. The quantity of specific bio-waste sludges both from the industrial processes as from the pre treatment of the crop or from the waste water treatment, are specific for the food industry.
- Animal by-products
- Waste from waste treatment industry, pre-treated bio-waste

Following, more general, trends can be assumed, regarding future evolutions for industrial bio-waste:

- Waste treatment capacity will not be developed merely for household waste or municipal waste, or for bio-waste from these sources. Waste treatment capacity, either composting or anaerobic digestion for source separated waste or MBT, incineration or landfill for mixed waste, will be developed for waste which is equal or comparable in nature and composition, disregarding its origin from households or from industry. A major trend will also be the growing co-digestion of (household and industrial) biowaste with manure (not always considered waste) and with energy crops or other sources of biomass.
- Waste from industrial sources is often more uniform in its composition and less polluted, and available in larger quantities. It is therefore usually more suitable for recycling activities and if recycling is economically beneficial it can compete on the available recycling capacity with more heterogeneous and more mixed post consumer fractions of bio-waste.

- Waste prevention can be more effective than for household waste because it tackles a smaller number of producers producing a larger quantity of waste. The Pareto Principle can be applied, where the largest effect is obtained by the first measures on/by a small group of generators.[35] Of course this is only the case where bio-waste is generated by large dominating companies, and not where the generation of bio-waste is dispersed over a large population of small SME's.
- Waste collection and waste treatment markets are becoming more and more integrated and consolidated. Only a few large market players remain, which offer more and more integrated services to their industrial clients, taking care of all or most of their waste fractions. This situation differs from the way in which household bio-waste collection and treatment can be organised, where bio-waste is collected and sent to an (often municipal or public) composting plant. Industrial bio-waste will be handed over to an integrated waste collection and treatment company, and it will decide, often without consulting the original producer, if the waste will be treated in their own recycling or energy recovery infrastructure, or traded to external treatment installations, like companies preparing energy mixed for AD, or pre-treated and exported. Market trends and fluctuations in the value of the collected material will become increasingly important in a more flexible market for industrial wastes.

Recycling targets for industrial waste can thus be set much higher than for municipal waste.

Therefore, setting common recycling targets to municipal and industrial waste would not be the best approach as it does not respect the nuances between the different sources.

We thus propose to use separate recycling targets for both waste streams.

For industrial waste it will be important to be very explicit of what constitutes waste production, reuse, recycling and disposal. Inconsistencies in data are likely to lead to issues, for example concerning material production by-products that may be applied directly to land or be sent for pet food etc. The risk is that it could be possible to meet recycling targets by creating 'waste' from the utilisation of process by-products. This issue is, however, addressed in existing guidance by the Commission.

A major problem on industrial bio-waste is the lacking information or the poor quality of the available information. The recent data on industrial waste prepared by IPTS[36], as used in the preparation of the Green Paper, cannot be used because of their poor quality and inconsistency with other data sources. The data from EUROSTAT, collected in the frame of the Waste Statistics Regulation 2150/2002/EC, are incomplete and, if existing, refer to categories and definitions not compatible with the definition of bio-waste as included in the Waste Framework Directive. The data reported by the Member States in the data collection exercise of this study is very limited.

For this reason, the Commission services have agreed with the proposal of the project team to restrain from constructing a baseline sceanri on the generation and treatment of industrial bio-waste in EU-27 and its Member States. Based on the available information from Lithuania and from the Flemish Region of Belgium, cases are presented in Annex D to this report. They illustrate the actual production and treatment of bio-waste, the division of the generation over different industrial sectors and the distinctions in nature and composition of different types of industrial bio-waste. The Annex is dedicated to a comparison of the bio-waste definition and the available data from Eurostat.

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[35] The Pareto Principle states that, for many events, roughly 80% of the effects come from 20% of the causes.
[36] Study of waste streams and secondary materials in the EU, IPTS, November 2007

7 Fundamental Principles

7.1 Scope and Approach

In order to understand fully the results from the appraisal of Scenarios, it is important to understand some of the underlying factors that influence the costs and the benefits of different approaches to biowaste management. The intention is to highlight which factors influence what might be the most suitable systems to operate in the different Member States. In order to do this, we use cost benefit analysis modelling in a more distilled manner than is subsequently shown within the full impact assessment used for the full analysis of policy proposals. The rationale is that interpreting results from a macro level policy assessment is not possible without understanding the underlying principles. Effectively, the lessons that would one might seek to learn from the policy assessment would be masked by the interplay of a more significant number of issues.

Our view is that it is hard to know how to design policies that lead to the best outcomes without insight into how individual technologies and waste management practices perform under particular circumstances, and without a thorough understanding of the principles that affect the costs and benefits one way or the other. As such, by investigating the sensitive factors within the cost benefit modelling we are able to demonstrate points of principle for how policy can best be developed.

The analyses undertaken within the following sections investigate distinct elements of the modelling and demonstrate specific points of principle, thereby highlighting the reasons for variation from country to country. Country-specific assessments have merit as there are likely to be factors at play on a national scale that can impact the assessment one way or the other. For instance, it may be expected that where a country's electricity supply is principally supplied from low impact energy sources (renewables) then there is less benefit to be gained from generating electricity from biogas or incineration, and other solutions may take priority (such as use of biomethane for transport fuel). Similarly, if one assesses costs through the private metric (in which the effects of taxes and subsidies are included), then some countries' policy mechanisms (for example, landfill / incineration taxes, or policies to promote renewable energy generation) may tend to influence the costs of some options more than others.

The methodology followed, therefore, is to model, for individual Member States, 'tonne for tonne' comparisons for the different approaches to biowaste management, highlighting the financial costs and the environmental costs and benefits, and the reasons for variation therein.

The discussion covers:

- Food waste prevention;
- Home composting;
- Collection;
- Composting and AD (including electricity production, vehicle fuel substitution and gas to grid variants);
- Incineration (including electricity only and combined heat and power (CHP) variants);
- Landfill;
- Switching from incineration to AD; and

- Switching from incineration to AD where the capital costs are assumed to be zero (as a proxy for the case where costs in incineration are 'sunk').'

The knowledge gained through these assessments will lead to the recommendation of solutions which would be most appropriate for individual Member States in their specific circumstances. We also conduct analysis, for the purposes of sensitivity, of what happens in situations where countries have, or would have if they separately collected biowaste, surplus incineration capacity. In these situations, an extreme view would be that the capital costs are effectively sunk, and the costs diminished accordingly. On the other hand, the potential for cross-border movements exists at facilities which are classified as recovery facilities (so that capacity should not necessarily be seen as an exclusively national issue).

7.1.1 Additional note on Marginal Electricity Sources

It is important to note that in the context of this study, what is typically being assessed is the effect of systems which are likely to contribute to a still growing demand for electricity across the EU. It is also important to note that most energy generation from waste is not of an intermittent variety (though this could be the case where solid recovered fuel (SRF) is concerned, or where the source of electricity is biogas, which could be stored). In most cases, the generation of electricity will contribute to the base load. Consequently, it is not appropriate to consider the 'marginal' source in the sense of 'the order of despatch' of electricity from different sources. Rather, since the contribution is to base load, and since demand for electricity is growing, then to the extent that we are looking at new facilities in future, the marginal source ought to be the other main sources of electricity contributing to additional base load in future (this argument might be less strong for existing incinerators / AD facilities/ MBT facilities which generate electricity). It seems reasonable to expect that in future, these will have a carbon intensity which is below the average level (otherwise, the average carbon intensity will not decline).

Although in principle the EU market for electricity is becoming more integrated, in practice, trading of electricity is somewhat fragmented, and national infrastructure is quite dominant in terms of the national supply. As such, it does not appear to make sense to use the same 'avoided source' for each country of the EU for the simple reason that the market is not yet so integrated. In situations where markets were well integrated, it might make sense to assume that the marginal source was the same in all countries.

Given the above, it would have been appropriate to assume that the electricity source being avoided was the marginal source contributing to base load. In practice, this is difficult to determine for each of 27 EU Member States. We have, therefore, assumed that the avoided source of electricity is the average mix within the country concerned.

7.1.2 Key Aspects of Methodology

We have modelled financial costs under a 'private' and a 'social' cost metric. The cost of capital used for the private cost metric ranges from 10% to 15%, depending on the process. This reflects a Weighted Average Cost of Capital (WACC) valuing the opportunity cost of capital investments. The private cost metric also includes the relevant taxes, subsidies, support mechanisms that apply to the management of waste.

We have used the European Commission standard discount rate for impact assessments for the social metric at 4%.[37] Under the social metric, the discount rate is used as the cost

[37] European Commission (2009) Impact Assessment Guidelines, 15th January 2009. This discount rate is expressed in real terms, taking account of inflation, and is therefore applied to costs and benefits expressed in constant prices. The rate broadly corresponds to the average real yield on longer-term government debt in the

of capital. The calculation of financial costs under the social metric does not include the effect of taxes and subsidies. This is explained more fully in Annex E.

The key point to note regarding the variation in total cost for the Member States is that it is related to the method we have used to adjust the capital and operational expenditure across the EU. We have allocated a percentage of labour cost related to the assumed operational and capital expenditure. The labour cost per country has been varied based on an index of labour rates[38]. A comprehensive explanation of this methodology is available in Annex E.

It is important to note that these costs are not identical to gate fees. Financial costs as measured by the private metric should resemble gate fees more closely than the financial costs under the social metric.

7.2 Food Waste Prevention

On the basis of the data gathered from the Member States, it is clear that food waste constitutes a significant proportion of the household and municipal waste streams. In very few cases is the proportion of food waste less than 20% of the total.

There is growing recognition that much of this food waste may be avoidable. However, no systematic investigations across the whole of Europe have been undertaken as to the possible reasons why food is wasted, and the degree to which it might be avoidable. The most significant study has been that undertaken is on behalf of the UK body, the Waste and Resources Action Programme (WRAP). The key results from this study are truly staggering. They suggest that in the UK:[39]

- 6.7 million tonnes of food waste are generated by households each year
- The value of this waste is approximately €19 billion, and amounts to approximately one third of all food purchased
- Of the food waste generated:
    - that which was 't*truly* unavoidable' was only 19% of the total
    - that which was '*avoidable' was estimated at 61% of the total*, with a value of approximately €12 billion. 8% of avoidable waste was '*in date*' (i.e. not past its sell-by date) when it is disposed of;
    - That which was '*possibly avoidable*' was 20%of the total;
- Major items of avoidable food waste included:
    - 359,000 tonnes of potatoes, of which 177,000 tonnes were whole or untouched
    - 190,000 tonnes of apples, of which 179,000 were whole or untouched
    - 45% of all salads purchased; and
    - 1.2 million tonnes of food still untouched and in its packaging

We are aware that similar concerns regarding food waste are being highlighted in Austria, Germany and Belgium.

EU over a period since the early 1980s. For impacts occurring more than 30 years in the future, the Guidelines state that a declining discount rate could be used for sensitivity analysis if this can be justified in the particular context.

[38] Poyry (2008) *Compliance Costs for Meeting the 20% Renewable Energy Target in 2020: A Report to the Department for Business, Enterprise and Regulatory Reform,* available at: http://www.berr.gov.uk/files/file45238.pdf

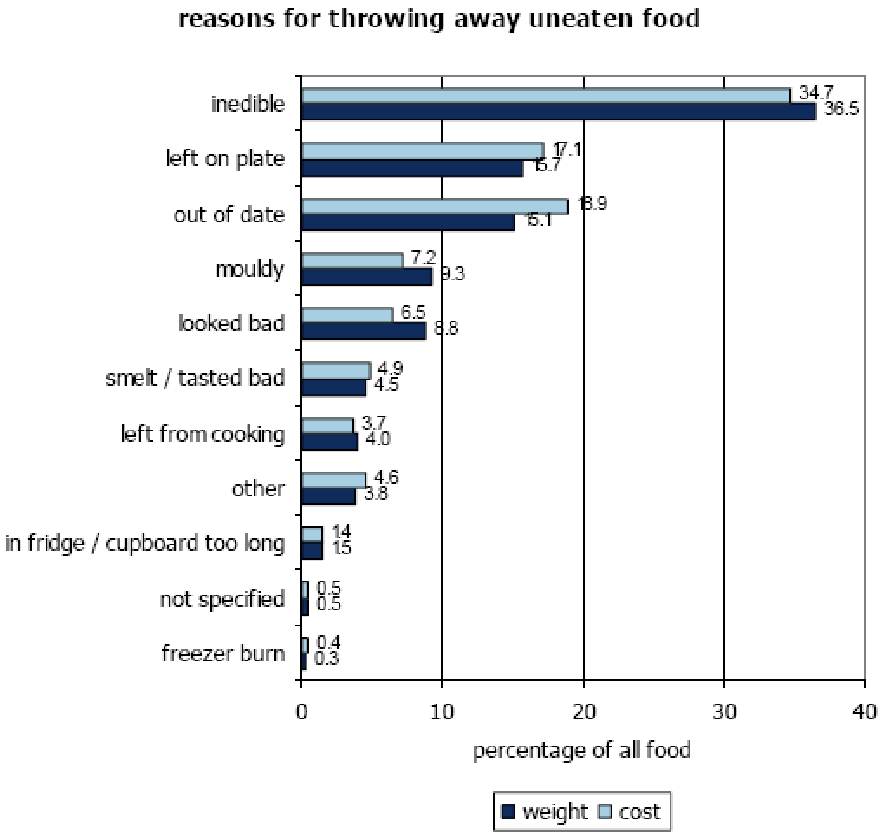
[39] Exodus Market Research (2008) *The Food We Waste,* Report for WRAP, April 2008.

It seems reasonable to believe that neither the nature of food wastes, nor the reasons for the waste arising, will be the same across the EU-27. Nonetheless, some common themes might be expected. In the WRAP study, the principle reasons for food being wasted were identified as:

1. left on the plate after a meal (1,225,700 tonnes worth €4.1 billion);
2. passed its date (808,000 tonnes worth €2.8 billion);
3. looked, smelt or tasted bad (750,500 tonnes worth €2.3 billion);
4. went mouldy (465,700 tonnes worth €1.25 billion); and
5. left over from cooking (360,600 tonnes worth €1.00 billion).

One might speculate that in some countries and in some situations, food left on the plate or left over from cooking would be used to feed livestock, but this is less likely in urban situations. The other reasons relate to food purchasing and food storage issues. These are likely to be of relevance in many countries and situations, though clearly, the levels of consumption, and hence, arguably of waste, will be influenced by income levels. At the same time, there is some suggestion that the composition of food waste in household waste in different countries is such that it is a higher proportion in countries which have lower incomes and less waste. As such, it might be argued that food waste quantities are relatively constant across countries, but the reasons for wastage, and the types of food waste, would be expected to vary.

Figure 7-1: Reasons for Throwing Away Uneaten Food in the UK



Source: Exodus Market Research (2008) *The Food We Waste,* Report for WRAP, April 2008.

7.2.1 Financial Costs

In respect of food waste prevention, the issue is not so much one of costs, but what is required to encourage households to avail themselves of the potential savings. As Table 7-1 shows, in the UK, households typically waste around £400 (or around €500) each year on *avoidable* food waste. In addition, there are clearly costs to municipalities (and hence, to households through the payment for the service) of collecting and treating / disposing of this waste.

What seems to be necessary is a concerted campaign to reduce the level of wasted food. WRAP has launched, in the UK, a highly regarded campaign entitled 'Love Food, Hate Waste'. It is difficult to measure the success of these campaigns in terms of outcomes. Various factors are at play. However, it goes without saying that it would be very difficult to imagine a public campaign exceeding the value of the avoidable food waste. Indeed, it would be difficult to imagine such a campaign exceeding the value of 10% of the avoidable food waste. If, therefore, seems reasonable to argue that there may be a role for concerted education campaigns which seek to inform households of good practice in food storage.

Table 7-1: Value of Avoidable Waste According to Household Type in the UK

Household Type	Value of Avoidable Waste (per year)	Value of Avoidable Waste (per week)
Single-occupancy households	£250	£4.80
Shared households of unrelated adults	£520	£10.00
Households of related adults	£380	£7.30
Households of related adults with children	£610	£11.70
Other2 (Scotland only)	£250	£4.80
Average England household	£420	£8.10
Average Scotland household	£410	£7.90
Average Wales household	£420	£8.10
Average Northern Ireland household	£440	£8.50
Average UK household	£420	£8.10

Source: Exodus Market Research (2008) The Food We Waste, Report for WRAP, April 2008

7.2.2 Links to Food Waste Collections

There may have been a number of studies conducted locally within the EU concerning the possible linkages between collection systems and waste generation. There are very good reasons to believe that the way in which biowaste is collected will influence the quantities of waste generated. Much of this discussion has focussed upon the effect of free garden waste collections on the collected quantity of garden waste (see Section 7.4.1 below). The potential consequences of the design of the collection for food waste generation appears to have been less well studied.

It is well known that schemes variously known in different countries as variable charging, pay-by-use, DIFTAR, differential charging and pay-as-you-throw, in which households are charged either by frequency, volume, or weight, or a combination of more than one of these, can lead to:

- A waste prevention effect; and
- An increase in the proportion of the remainder which is recycled.

In a growing number of countries – for example, Austria, Belgium, Czech Republic, Germany, Ireland, Italy, Netherlands – this type of approach to incentivising households to change behaviour is becoming, or has already become, the norm.

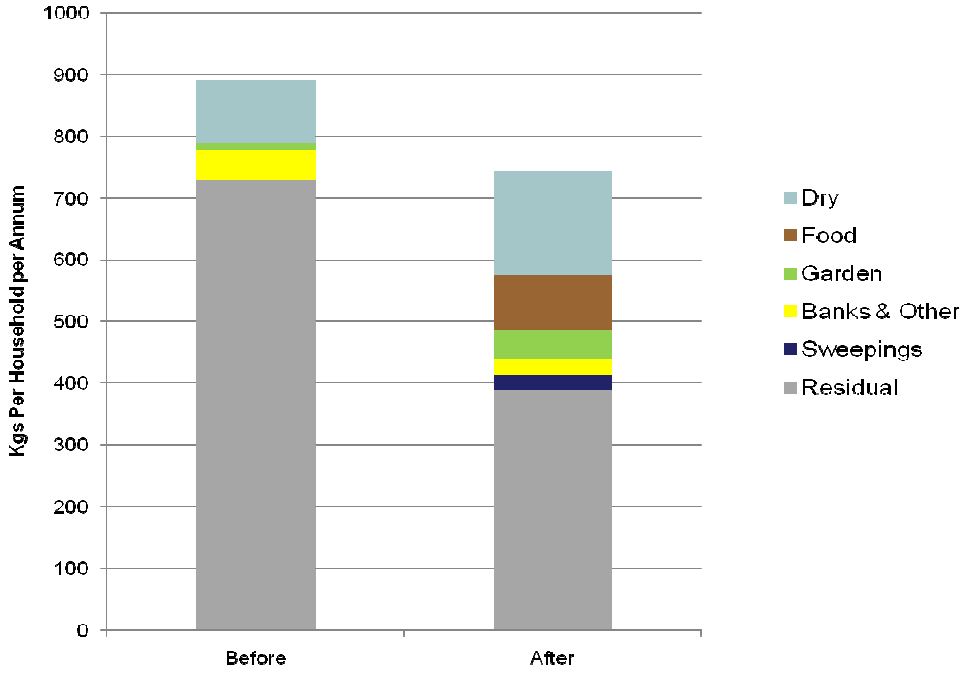
There are some general rules of thumb regarding the use of this type of system, notably, that the services which are on offer to households, in terms of convenient alternatives to the refuse system, are broad in scope, and typically, includes some form of biowaste collection. The explanations for the waste prevention effect are not entirely clear, but it seems reasonable to suggest that where such schemes are used, and where there is a collection service for kitchen waste in place, that some of the prevention effect relates to a reduction in food waste prevention.

Indeed, some evidence of such an effect appears to come from UK systems where food waste collections have been introduced, even where no charges are applied. In the case shown in Figure 7-2 below, the system switch involves introducing a food waste collection system alongside a reduction in collection frequency for refuse from fortnightly to weekly.

As well as the system giving rise to a significant uplift in dry recycling, an important subsidiary benefit, the introduction of food waste collections and the drop in refuse frequency gives rise to a significant reduction in waste generation (almost 20%). In this system, on the basis of 'before and after' composition analysis, the scheme estimated a reduction in food waste generation of 25%.

These effects – the link between the collection system and the generation of waste – are still relatively poorly understood, but it seems far from fanciful to imagine that as households are asked to sort kitchen waste from their refuse, they would become more aware of exactly the form of wastage identified by the WRAP study discussed above.

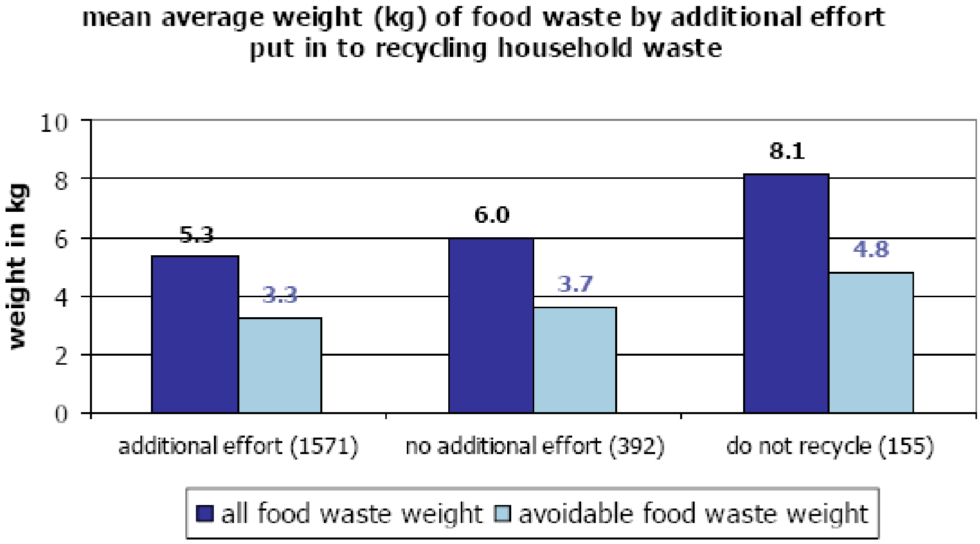
Figure 7-2: Effects of Introducing Weekly Food Waste Collections and Fortnightly Refuse Collection in Somerset, UK



Source: May Gurney

Other evidence comes from survey work. Efforts to recycle appear to be linked to reduced quantities of food waste generation, suggesting that these may be linked behaviours (see Figure 7-3).

Figure 7-3: Links between Food Waste generation and Efforts Made to Recycle in the UK



Source: Exodus Market Research (2008) The Food We Waste, Report for WRAP, April 2008

This highlights the fact that it is not generally correct to compare systems on the basis that the mass flows within them are the same. The collection system will influence the generation of waste. Free garden waste collections can increase quantities collected. On the other hand, food waste collections – especially where they are accompanied by reductions in refuse collection frequency (which acts as a constraint on waste generation) – may lead to a reduction in food waste generation in the first place. The significance of this is explored below in respect of environmental impacts.

7.2.3 Environmental Impacts

The EU Environmental Impact of Products study suggested that the food, drink tobacco and narcotics group of products accounted for 20-30% of the environmental impact of consumption for all products (this is higher for eutrophication). The highest proportion of impacts come from meat and meat products, and from dairy products.[40] A recent review of the literature also highlights potentially significant environmental impacts from food production. Other literature tends to support the view that rankings follow this pattern, though they highlight some exceptions to these rules (for example, it may be that where rice cultivation leads to methane releases from rice fields, climate change impacts are very significant).[41]

Significantly, WRAP estimates that for every tonne of food waste avoided, around 4.5 tonnes of CO_2 equ. are avoided.[42] This is clearly highly significant. The monetised benefits from the avoided greenhouse gas emissions alone would amount to around €121.5 per tonne of food waste avoided.

In our modelling we have assumed an impact of this magnitude. This does not include, therefore, any commensurate benefits associated with non-GHG air emissions.

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[40] A. Tukker et al (2005) *Environmental Impacts of Products (EIPRO) Analysis of the Life Cycle Environmental Impacts Related to the Total Final Consumption of the EU-25*, IPTS/ESTO, 2005.

[41] See, for example, Danish Institute of Agricultural Sciences (2003) *Life Cycle Assessment in the Agri-food Sector*, Proceedings from the 4th International Conference, October 6-8, 2003, Bygholm, Denmark; Manchester Business School (2006) *Environmental Impacts of Food Production and Consumption*, Report for Defra, December 2006.

[42] Barthel, M, "The Importance of Tackling Food Waste", WRAP http://www.wrap.org.uk/downloads/Mark_Barthel_-_Introduction_context.4b08ccfa.5939.pdf .

7.3 Home Composting

There have been relatively few thorough evaluations of the costs and benefits of home composting. Environmental emissions were assessed in a UK study and also in a study by Ifeu. Some studies have assessed the greenhouse gas emissions from different composting systems, including home composting. Some of the key issues are discussed below.

7.3.1 Financial Costs

Consideration of the costs of home composting might depend upon the vantage point from which one is making the assessment. For example, from the perspective of a local authority, any outlay used to promote the uptake of home composting might be considered to reduce, at the margin, the costs of collection of waste and its treatment. The actual effect on costs will depend upon the quantitative reduction effect From the perspective of the household, there may be costs borne by the household, but there may also be cost savings if the nature of the system through which households are charged incentivises waste prevention.

The nature of home composting equipment will vary across households. In some more rural households, where the quantity of waste suitable for home composting may be significant, the most suitable equipment may be fairly simple to construct from waste materials such as pallets. In other areas, dedicated home composting equipment may be used for the purpose. These are usually (plastic) containers which are intended to form a barrier with the external environment. Home-made boxes may be just as, if not more, effective not least since they provide ready access to the material and make turning relatively straightforward.

Arguably, what is rather more important for the success of a home composting programme is the availability of advice and advisors to ensure the process is managed in such a manner that the material will be used by the makers of the compost. Eunomia remark that Vogt et al suggest that an essential assumption in the superior environmental performance of home composting was that the material should 'replace something', either bark or other soil improvers. Hence, the quality of the material produced is of some significance, and a mentality of 'landfill diversion anyhow', irrespective of the means or outcomes, would tend towards the production of materials of little or no utility.[43]

Eunomia estimated the costs of running promotional schemes through reference to work undertaken by WRAP seeking to promote home composting.[44] It was assumed that promotional activities were accompanied by support for the costs of home composting bins and / or boxes, and that compost counsellors were available to provide advice to users. In addition, home composting could be promoted at civic amenity sites / containerparks with displays and periodic demonstrations.[45] In costing full programmes,

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[43] R. Vogt, F. Knappe, J. Giegrich and A. Detzel (2002) Okobilanz Bioabfallverwertung Untershung zur Umweltvertraglichkeit von Systemen zur Verwertung von biologisch-organischen Abfallen. *Initiativen zum Umweltschutz* 52, Berlin; also Regine Vogt, Florian Knappe and Andreas Detzel (2001) *Environmental Evaluation of Systems for the Recovery of Biogenic Waste*, Proceedings from the ORBIT 2001 Conference; F. Knappe, R. Vogt and B. Franke (2004) Biowaste Management from an Ecological Perspective, in P. Lens, B. Hamelers, H. Hoitink and W. Bidlingmaier (2004) (eds.) *Resource Recovery and Reuse in Organic Solid Waste Management*, London: IWA Publishing, pp. 71-92.

[44] Eunomia (2007) *Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis*, A Final Report to WRAP.

[45] Member States use different terms to describe these sites, which are essentially sites where households may take a range of different, often including (but not only) more bulky, items for recycling or disposal.

whether or not the counsellors are paid or voluntary is important. In many successful schemes, such as that run by Vlaco in Flanders, the compost counsellors are voluntary, though they receive some in-kind benefits, and they are provided with a range of equipment and tools.

Eunomia focused only upon the costs of operating home composting schemes since in their analysis, the benefits (in terms of reduced collection costs to local authorities) were picked up in the remaining cost modelling. Costs estimated are shown in Table 7-2.[46] The cost assumptions assumed that the local authority partly subsidized the sales of home composting containers. If one relaxes this assumption, then the costs of the bins increase, but with no net increase in the cost to society. The actual cost at which these are made available to households will depend upon the way in which they are purchased, but bulk purchases should keep costs to levels of the order €25 per system. At these levels, costs would rise from £1.85 in 2006 sterling, or around €2.50 in 2009 Euros, to around €4.15 per household.

Table 7-2: Financial Costs of Home Composting (£/hhld)

Cost Item	Cost (per bin)
Marketing, literature & support	£5
Net cost of bin (after sales revenue)	£2.50
Delivery / Distribution (home delivery) & Storage Costs	£11
Overall	£18.50
Assumed Life of Bin	10 years
Annualised Cost of Bin (per bin issued)	£1.85

Source: Eunomia (2007) *Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis*, A Final Report to WRAP

This figure is somewhat lower than the figures used by GESPER, who estimate costs of between €11 and €18 per household. Key assumptions relate to the assumed life of the container, and the degree to which the costs of support are deemed to arise annually or not. The figure of €18 per household on an annualized basis is, however, somewhat difficult to understand. Given the low annualized cost of the equipment, such a cost would only be arrived at assuming a very costly plan for informing and assisting citizens.

The degree to which home composting leads to a reduction in costs for local authorities depends in part upon the impact of home composting on the costs of the collection system provided to households. WRAP estimates that the effect on collection, through household waste recycling centres (HWRCs, akin to containerparks, or civic amenity sites) and through doorstep collections is as shown in Figure 7-4. The average figure suggests that home composting may lead to the management of as much as 220kg per household. GESPER suggests figures of 170kg per household, which is close to the WRAP figure for gardens of less than 200m^2, whilst Vlaco suggests a figure of 50kg per inhabitant for participating households in 2002.[47]

If one combines the adjusted Eunomia estimate and the lower end figure from GESPER (€4.15 - €11 per household) with the range of quantities being diverted (125 –

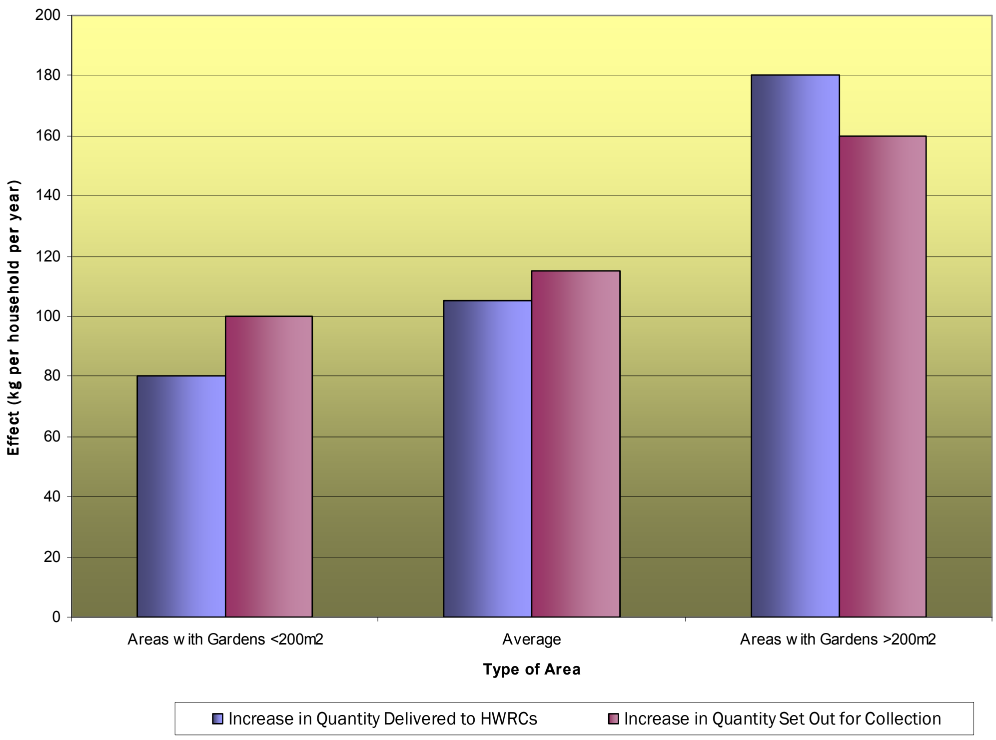
[46] The costs assume that one does not include the costs of time spent in the process. There is an ongoing debate as to whether, and if so, how, this should be included in CBAs around recycling and composting. For a discussion, see, for example, Eunomia (2006) *Impact of Unit-based Waste Collection Charges*, OECD Report ENV/EPOC/WGWPR(2005)10/FINAL; Smith, S., (2005) *Analytical Framework for Evaluating Costs and Benefits of EPR Programmes*, Report for OECD Working Group on Waste Prevention and Recycling, ENV/EPOC/WGWPR(2005)6/FINAL, http://appli1.oecd.org/olis/2005doc.nsf/linkto/env-epocwgwpr(2005)6-final.

[47] Jan Buysse (u.d.) *Lessons from the Flanders Composting Experience*.

220kg/household), then a cost varying between €19 - €50 per tonne of waste being home composted is obtained.

The costs which home composting helps to avoid depend upon the exact detail of the collection system being used, and the way in which home composting affects participation in the collection scheme. There may be circumstances, for example, where a household is home composting, but where it continues to participate in a biowaste collection scheme. The nature of its participation will, therefore, determine the influence upon the costs of collection, whilst the collection scheme itself will determine the avoided cost of treating the material which is home composted (would the material otherwise have been collected separately for composting / anaerobic digestion, or would it have been collected as residual waste, and if so, would it have been landfilled, or incinerated, or treated through other means?). Other things being equal, however, one would expect costs to fall by *at least* the avoided cost of the treatment of the material. In other words, as long as the avoided costs of treating the waste which is being home composted (i.e. the savings from not treating the waste) are greater than €19-50 per tonne, then the costs of home composting will, in the round, be negative. Interestingly, the distribution of these costs and, especially, the savings, is likely to depend upon how the collection service is funded, and the degree to which the household itself benefits (through avoided payments) from the reduced cost of the service.

Figure 7-4: Effect of Participation in Home Composting on Waste Collected at HWRCs and through Doorstep Collections (average kg per household participating).



Source: WRAP.

7.3.2 Environmental Impact

One UK study looked at emissions from a number of home composting sites.[48] The level of emissions was found to be very low for those gases for which measurements were sought. It is not possible to translate the estimates of emissions into emissions 'per tonne' of home composted material. The emphasis was on the measurement of concentrations rather than emissions per tonne.

The experiment was carried out at 12 volunteer home compost sites. The authors state:

The analysis for methane has resulted in only two reading at one of the sites (out of 112 readings) being above the detection limit of the equipment used (0.1 %) and further tests are underway with more sensitive equipment to ascertain levels more precisely. The CO2 readings have measured values 32 times out of the 112 readings. It is likely that the methane emitted is very low and does not pose any significant environmental contribution to global climate change, but results from the more accurate equipment may shed some light on this.

Commenting on the study, Eunomia suggest that the low rate of emissions from home composting and the ratio of the surface area to the volume make it possible that emissions from home compost heaps are likely to be subject to 'biofilter-type' abatement as the outer layers act to abate, at least to some extent, the emissions from the degrading material. However, Amlinger, Cuhls and Peyr suggest that biofilters might not be especially efficient at removing methane, so this suggestion might not be correct. Indeed, the same authors suggest that home compost heaps may generate rather more greenhouse gases than other modes of composting (76 and 187 kg CO_2 equivalent for two experiments), though they suggest the upper value obtained is questionable and might have been a result of measurement errors. They also suggest that even at small sites, infrequent turning and a too high proportion of food in the feedstock may be factors leading to higher emission factors for CH_4 and N_2O.[49]

The work of Vogt et al made clear that an essential assumption in the superior environmental performance of home composting was that the material should 'replace something', either bark or other soil improvers. Hence, it is clearly important for the analysis that those who do compost at home also make good use of the resultant material.

Eunomia suggest that home composting might not deliver the same environmental benefits as anaerobic digestion. They suggest, however, that in the overall analysis of costs and benefits, the financial cost savings are likely to outweigh the reduced benefits from not collecting waste for anaerobic digestion. They suggest, therefore, that net welfare gains are to be made from home composting.

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[48] P. A. Wheeler and J. Parfitt (u.d.) *Life Cycle Assessment of Home Composting*, Environment Agency R&D Report CLO329.

[49] F. Amlinger, C. Cuhls and S. Peyr (2008) Green House Gas Emissions from Composting and Mechaniscal Biological Treatment, *Waste Management Research*, Vol.26 (1) pp. 47-60.

7.4 Collection

Before the costs and benefits associated with different methods of collection can be interrogated, a number of general principles need to be established. The collection systems themselves influence waste production and impact on the viability of particular treatment systems. Such factors are defined in Section 7.4.1 before the costs and environmental impacts are appraised in Section 7.4.2 and 7.4.3 respectively.

7.4.1 Impacts Associated with the Method of Collection

7.4.1.1 Collection System Impacts on Waste Generation

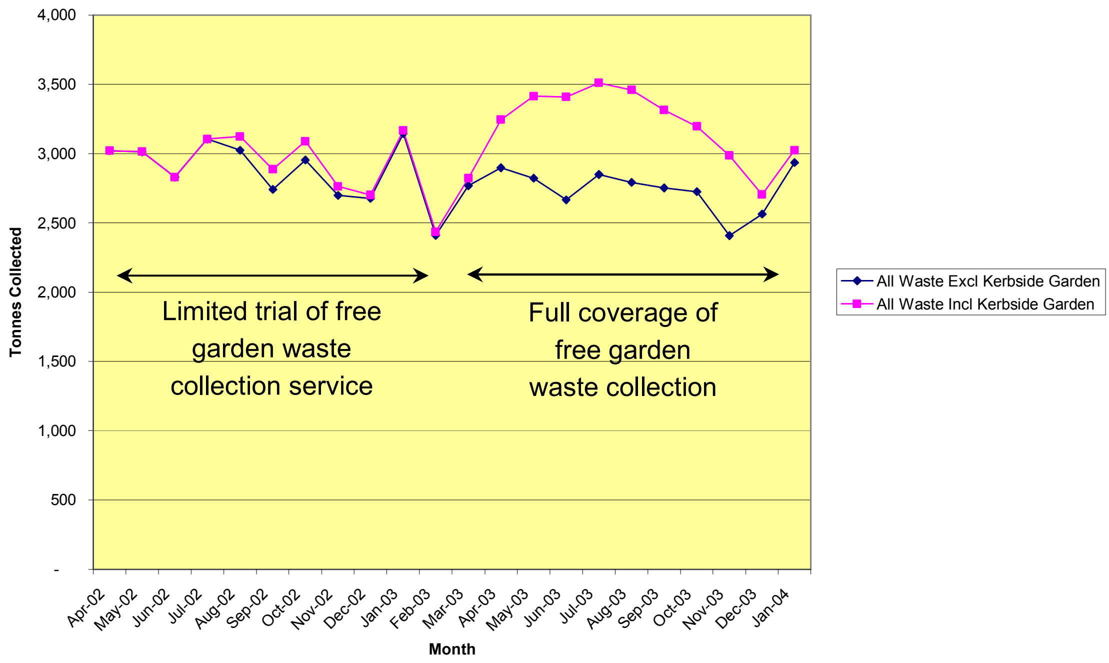
The nature of waste collection systems and provision of centralised facilities has significant impact on how waste is, and can be, managed. Where collection services are limited to a single stream collection, this forces all waste materials to be combined and limits the treatment methods to only those designed to handle mixed waste. Although systems exist that attempt to segregate materials from mixed waste, product quality is always an issue and the subsequent applications for outputs (and the associated benefits) are more limited. Multi stream collections allow much more bespoke processing routes. This is as true for biowaste as it is for dry recyclables. Keeping food separate from garden waste allows optimisation of the processing routes for each biowaste element.

The treatment options for commingled biowaste (food and garden waste combined at the point of collection) are limited due to the nature of the material mix. An issue is that the relative concentrations of food to garden waste are not controllable at the processing stage; the ratio is effectively set by the householder. Furthermore this ratio will fluctuate through the course of each year. Data in Figure 7-5 shows two years of total waste arisings tonnage data for a UK authority where a free garden waste collection service was put in place. The difference between the two lines of data represents the quantity of garden waste that was separately collected. The service was trialled for a limited number of households in the first year before being provided to all households in the second year. Several important characteristics of garden waste generation and the impacts of collection services can be observed in the data:

- Firstly, the impact of providing a new and free collection service is seen to increase total waste arisings; this can largely be attributed to a reduction in home composting and a potential encouragement of harvesting a latent garden waste fraction (i.e. the new outlet for garden waste brings additional material into the waste management system that was either not created or previously managed at source – material from hedge cutting for instance);
- Secondly, the fact that the blue line falls slightly in the second year indicates that some garden waste is diverted from residual waste collection in addition to material that may have been diverted using other, already existing management options (such as recycling). However, it is arguable in this case that much of the reason for this material being disposed in the first year was due to the zero cost nature of UK residual waste collection services. This type of activity is more effectively discouraged in pay-as-you-throw systems;
- Thirdly, and most importantly, the data for the second year shows the seasonal nature of garden waste generation. In the UK the growing season and active gardening period runs through the warm spring and summer months with generation falling nearly to zero through the winter. Although the seasonal pattern

may differ from country to country depending on climatic conditions, a seasonality effect can be expected to occur in every member state.

Figure 7-5: Waste Arisings Before and After Free Garden Waste Collection in a UK Municipalility



Although food waste does not tend to exhibit observable seasonality, the variable pattern in garden waste generation has implications for the treatment systems operated. This is especially the case where food and garden waste are collected together. Seasonal garden waste generation will mean that the relative concentration of food to garden waste will vary throughout the year. This can make it difficult to optimise the treatment system and will have a potentially negative impact on treatment economics. We explore this issue further in the following subsection.

Another important point is that where garden waste collections are offered free of charge, they tend to undermine the potential for home composting. As far as home composting promotion is concerned, it is suggested here that it is increasingly difficult to achieve results where free garden waste collections are put in place. The convenience that free garden waste collections offer to households in terms of depositing garden waste tends to discourage dealing with the material at home through home composting.

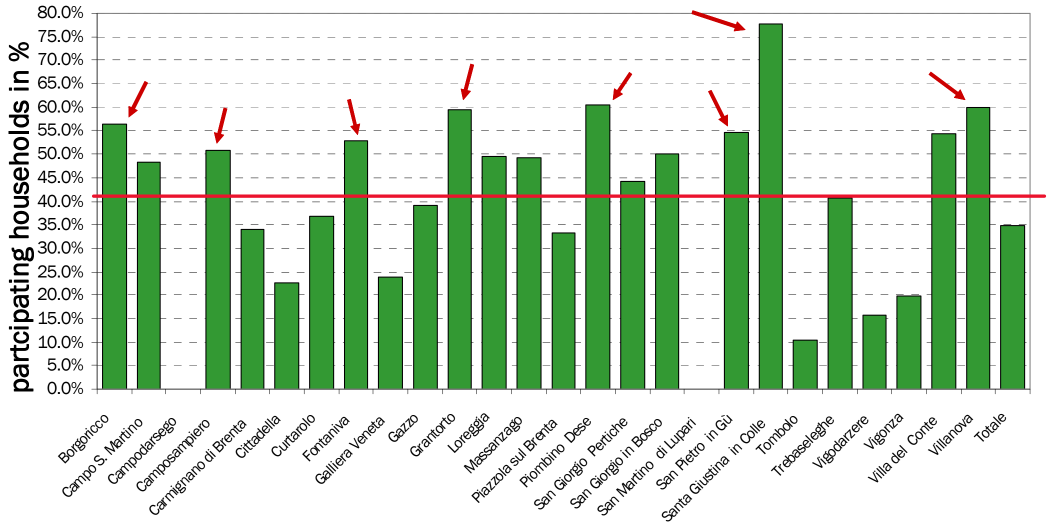
European experience also supports the view that it is extremely difficult to constrain delivery of garden wastes through promotion of home composting where the garden waste collection is free at the point of delivery. Rather, the option of charging for garden waste collections (typically, with higher charges for collection of refuse) needs to be available in order to further encourage home composting if there is to be a doorstep collection of garden waste at all.[50]

Figure 7-6 below illustrate the outcomes of a survey of home composting in Italian municipalities, showing the rates of home composting in those households with and

[50] These comments are based upon a number of study visits to European municipalities, as well as on reviews of performance of systems where charging of householders takes place. The 'minimisation' effect of charging systems is often explained principally through reference to an increase in home composting / mulching. See, for example, Eunomia (2003) *Waste Collection: To Charge or Not to Charge?* Report for IWM(EB).

without (shown with arrows) free garden waste collections. Participation rates are generally higher in municipalities without free garden waste collections.

Figure 7-6: Plot Showing Home Composting Participation Rates in Italian Municipalities With and Without Free Garden Waste Collection (municipalities with garden waste collection)



7.4.1.2 Collection System Implications for Treatment Options

When collected for centralised management, food waste containing products from animal origin is subject to the EU Animal By-Products Regulation (1774/2002). This regulation attempts to mitigate the risks associated with animal by-products re-entering the food chain. Garden waste (when collected separately or together with only fruit and vegetable waste) does not fall within this regulation and is routinely treated in simple open air composting facilities which tend to have low infrastructure and operating costs. Household food waste (that includes or potentially includes waste from animal origin) is subject to the rules affecting catering waste under the EU Regulation. In this case, composting/anaerobic digestion facilities must demonstrate that they achieve levels of hygienisation of the feedstock material. In most countries, the resulting processing standards (and capital infrastructure investments) are different than for garden waste, and costs are increased. It is not the case, however, that all EU Member States require the treatment of food waste in fully enclosed systems, though many do.

Where food waste is separately collected, it is increasingly treated in in-vessel (or enclosed) facilities. Where the facility is a composting plant, the food waste has to be bulked with a quantity of garden waste. An excessively high proportion of food waste in an in-vessel facility leads to settlement and the development of anaerobic conditions, odour and methane generation etc. Garden waste provides the structure needed to maintain air pathways through the material to limit this effect. As such, in-vessel composting is often coupled with commingled food and garden waste collections. However, the varying ratio of food to garden waste associated with commingled collection can lead to processing problems (chiefly caused by the development of anaerobic conditions) away from the gardening and growing seasons of the year. Furthermore, plant capacities will have to be matched to peak levels of production, with throughput being reduced in winter months, consequently reducing the utilisation of the capital investment.

Although AD can process garden wastes, the readily fermentable fraction of this material is somewhat lower in comparison to food waste. Garden waste contains significantly higher proportions of lignin which does not break down under the anaerobic conditions in the digestor. As such, gas yields are reduced and residence times to achieve reasonable levels of methanation are higher. This technology therefore tends to be better suited to segregated food waste streams, though mixes are suitable for digesters capable of treating material with higher dry matter content. Damage caused to pipelines by stones and grit leading to increased servicing and downtime are also a common problem suffered by AD facilities accepting garden waste.

We may summarise that the technologies on the market for biological treatment of segregated biowaste fall into three basic categories:

- Open windrow composting – suitable for garden waste or garden, fruit and vegetable waste, or in some countries, garden waste and all catering waste from households;
- In-vessel composting – suitable for mixed food and garden waste. Process optimised through appropriate blending of food and garden wastes at the facility rather than commingled collection of biowastes;
- Anaerobic digestion – suitable for treating feedstocks with a high proportion of household food waste, but depending upon the technology, requiring some additional material.

7.4.2 Costs

The costs of implementing separate collection schemes are not straightforward to identify, not least because the options available are numerous with some being more expensive than others. It is important to note that there are numerous permutations available for biowaste collection. These include the following:

- Scope of materials collected can include any combination of garden waste and food waste, sometimes with cardboard included;
- Frequency of collections of the biowaste *and* the refuse can be such that frequencies are the same, or that the one is greater than the other. This affects the capture of the materials targeted, and the costs of the service;
- Vehicles used can include compactors or non compacting trucks with varying loads. The choice reflects the scope of materials (and their bulk density), the frequency of the collection, and the nature of the area being serviced;
- Containment methods may include bins, buckets, paper sacks, re-usable sacks, kitchen caddies and paper or starch-based liners, These affect the convenience of the service, and hence also, the capture, as well as being important cost items.

One of the issues with costing collection systems is that whether or not the system increases the cost of collection (and the system) depends upon the choice of *system*. Whether or not adding a collection of food waste, or garden waste, or kitchen and garden waste, will add cost to the collection system also depends on what the system was like before. Although this sounds obvious, it is an important point since introducing collections of biowaste offers the potential for optimisation of collection schemes, especially where putrescible material is targeted. Hence, although it is possible for additional collection services to result in a significant increase in net collection costs, typically, this is a consequence of poor design of the collection service, and failure to optimise the service.

In a cost benefit analysis, it is not appropriate to include, in the comparison, poorly designed separate collection systems if, for example, one intends to model well operated systems for treatment of biowaste / residual waste. Not all cost-benefit analyses have been carried out with such a level playing field. For example, one analysis in Denmark compared a system of refuse collection and incineration with a source separation scheme. The study suggested: [51]

The primary reason for recycling being more expensive than incineration is the necessary, but cost-intensive, dual collection of the household waste. Treatment itself is cheaper for recycling compared to incinerating.

In the analysis the extra cost of the dual collection is calculated on the basis of full-scale experiments/tests in several municipalities. The extra cost is about DKK 150 per household per year for single family houses and about DKK 110 per household per year for apartments. The extra cost must be below DKK 50 per household per year for single family houses and below DKK 20 per household per year for apartments in order to make anaerobic digestion more attractive than incineration.

The analysis by the EPA overlooks the possibilities for cost-optimisation in collection systems. The system modelled in the study – based on optical sorting of bags – is a poor approach for cost optimisation (though it may not be without uses in specific circumstances). Biowaste collections have a role to play in enabling a reduction in the frequency of refuse collections, but co-collection in sacks for subsequent optical sorting offers no such possibility (the refuse and biowaste collection frequencies are forced to be the same). Furthermore, the reject rates reported in the EPA study – 35% at the digestion plant and 15% at the compost plant - are so high as to make one realise that such an approach could hardly be considered the best one in the circumstances (for example, reject rates in other systems are regularly 5% and less). The approach also implied double handling of the waste for composting or digestion, again adding cost to the system, especially since the residual waste and biowaste facilities were not assumed to be co-located. The comparison made in the study, therefore, is effectively one between two systems, one of which – collection of refuse and incineration – is a high quality one, whereas the other – with additional collection of biowaste and either composting or digestion – is a very low quality and high cost system. Unsurprisingly, the study concluded that the high quality system was superior.

Interestingly, the EPA result suggests that the collection costs could be of the order €6/hhld more before the incineration route would be better than a separate collection system based where the material was treated through anaerobic digestion. As the discussion below suggests, in well designed schemes, this is frequently the case.

Our central assumptions are that the relevant collection systems can be introduced at zero additional cost. This is an important assumption and we set out the evidence for it below.

7.4.2.1 Collections of Food Waste

Successful segregation of the food waste fraction can facilitate a reduction in the required frequency of residual waste collections. This already happens in various municipalities in a number of countries, and is an especially important consideration in hotter climates,

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[51] Danish EPA (2003) *Skal husholdningernes madaffald brændes eller genanvendes? Samfundsøkonomisk analyse af øget genanvendelse af organisk dagrenovation* [Is the food waste from the households to be incinerated or recycled? A welfare-economic analysis of increased recycling of organic household waste.] Environmental project no. 814. http://www.mst.dk/udgiv/publikationer/2003/87-7972-685-2/html

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 food 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 residual municipal waste. For food waste collection, therefore, there is no reason why one has to incur the higher expense associated with such compacting vehicles. Non-compacting trucks can be used to collect the material, and again, this is already the case in municipalities in a number of European countries.

In the sub-sections that follow we pull together findings from specific instances in Europe where collection of food waste has been conducted with zero or negative net cost to the collection and treatment system as a whole.

7.4.2.1.1 Investigation of Collection Costs in Italy

In order to allow a comparison among different collection systems, the Research Group on Composting and Integrated Waste Management at Scuola Agraria del Parco di Monza led a survey on the costs of the different collection systems run in Italy, grouped mainly on the basis of the way food waste is collected separately (or not). The three system groups are as follows:

- **Traditional source separation of dry recyclables only**. This is based on the use of plastic bags or road containers (up to $3.3m^3$) for mixed MSW and source separation through road containers for paper, glass and plastics. The food waste is not sorted and remains with the residual waste. Residual waste therefore remains to a large degree fermentable (especially because the food waste element is further 'concentrated' in residual waste due to the sorting of paper, cardboard, glass, plastics, etc.) and has to be collected frequently;
- **Intensive source separation both for food waste and dry recyclables from communal collection points**. This is based on the use of 'road containers' (120-240 litres up to $3.3m^3$) for groups of houses rather than specific bins for each household individually. The collection of residual waste is also through road containers. This system is quite widely used in Central Italy (Emilia, Tuscany) and has also been the most common system, so far, in Spain (e.g. Cordoba, Catalunya);
- **Intensive source separation with collection of food and residual waste at the doorstep**. In this case each household has its own containers; collection is referred to as door-to-door (DtD). In general, high yields of dry recyclables are collected with a DtD system (usually paper and cardboard, due to the much higher capture per inhabitant than with road containers). This is the most common system in those Italian Municipalities and Provinces where the highest recycling rates are achieved (up to 70% in single Municipalities, and above 50% in Districts).

The outcomes of the survey follow.

7.4.2.1.2 Traditional Collection Systems

Compiled for the Eunomia cost benefit study for WRAP in 2007, Table 7-3 reports on the costs of the traditional collection system in Italy (commas within this report distinguishing thousands – not indicating decimal points).[52] The data shows that the total waste

[52] Eunomia, *Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis*, Report for WRAP, May 2007

management costs (including disposal) fluctuate widely because of the different disposal costs charged in different regions. Therefore, in order to evaluate the competitiveness and draw reliable conclusions it is necessary to focus on collection and transport costs, disregarding disposal costs.

The results also indicate that data expressed in cost per unit mass (ITL/kg) tends to penalise those municipalities whose specific (i.e. per household or per inhabitant) production of waste is lower. The average collection and transport costs of the three municipalities with waste arisings below 350 kg/inh/year is ITL 253/kg, while municipalities with more than 500 kg/inh/year have costs of ITL 134/kg. But in absolute terms, the latter must dispose of more waste, and overall waste management cost tends to be higher. Generally, in systems which capture higher quantities of material per pick up, the costs per unit weight can be expected to be lower.

The per capita cost of collection + transport (without disposal) for the traditional collection of mixed MSW averages ITL 66,000.

Table 7-3: Municipalities in Italy with a 'Traditional' Source Separation System Only for Dry Recyclables

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

NOTE: the average of the sums (average total cost) does not equal the sum of average values (average collection and transport + average disposal cost) due to the data not available.

7.4.2.1.3 Collection Systems with Source Separation of Food Waste

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

- Door to door (DtD) collection systems;
- Road container collection systems.

The study focused on mature experiences (schemes that had been run for at least two years), mainly concentrated in Northern Italy. Table 7-4 and Table 7-5 summarise the costs of the service. As previously noted, what matters is the average cost for collection + transport per inhabitant; we have highlighted this figure in both tables in bold font.

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

Table 7-4: Systems with Source Separation of Food Waste By Means of Road Containers

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

Table 7-5: Systems with Source Separation of Food Waste By Means of Road Containers

Municipality/ District	**Population**	**Average yearly MSW production (kg/inh/y)**	**Collection + transport cost (ITL/inh/ y)**	**Disposal cost (ITL/inh/y)**	**Total cost (ITL/inh/y)**	**Collection + transport cost (ITL/kg)**	**Total cost (ITL/kg)**
Venezia 4 District (4 Municipalities)	n.a.	321	53,733	31,558	85,291	167	266
Verona Province (7 Municipalities)	63,697	310	61,389	25,013	86,402	198	279
Padova 1 Basin (26 Municipalities)	206,000	322	52,500	25,182	77,682	163	241
Province Bergamo (7 Municipalities)	20,013	n.a.	45,821	62,954	108,775	n.a.	n.a.
Calcio	4,765	393	31,266	61,032	92,298	80	235
Caravaggio (after implementing the	14,181	n.a.	38,079	n.a.	n.a.	n.a.	n.a.

scheme)							
Cinisello B..	78,000	422	55,620	n.a.	n.a.	124	n.a.
Treviglio	25,294	457	n.a.	n.a.	158,310	n.a.	346
Cameri	9,567	382	n.a.	n.a.	83,521	n.a.	219
Castiglione	4,691	234	48,658	n.a.	n.a.	208	n.a.
Cupello	3,500	275	52,000	n.a.	n.a.	189	n.a.
AVERAGE		346	**48,401**	41,148	98,897	161	264

NOTE: the average of the sums (average total cost) does not equal the sum of average values (average collection and transport + average disposal cost) due to the data not available.

The traditional collection systems (road containers for the separation of dry recyclables only) shown in Table 7-3 show a higher cost per inhabitant than systems with a source segregation of food waste. This is partly due to higher collection frequencies in the case studies from Southern Italy (up to 6 times a week) that affect average costs, particularly since many case studies from Southern Italy are included in the table and the hotter climate necessitates more frequent collection of putrescible materials.

The most surprising outcome is that the average collection and transport costs (per inhabitant per year) tends to be lower in schemes where source segregation of food waste uses DtD systems, than where road containers are use, and lower also than systems where no food waste is collected. For many, this contradicts accepted wisdom due to the much higher number of pick-up points in DtD schemes.

A key feature of food waste collection is that it enables refuse to be collected less frequently as the problematic putrescible element is removed.

7.4.2.1.4 Collection Cost Case Study for Venezia 4, Italy

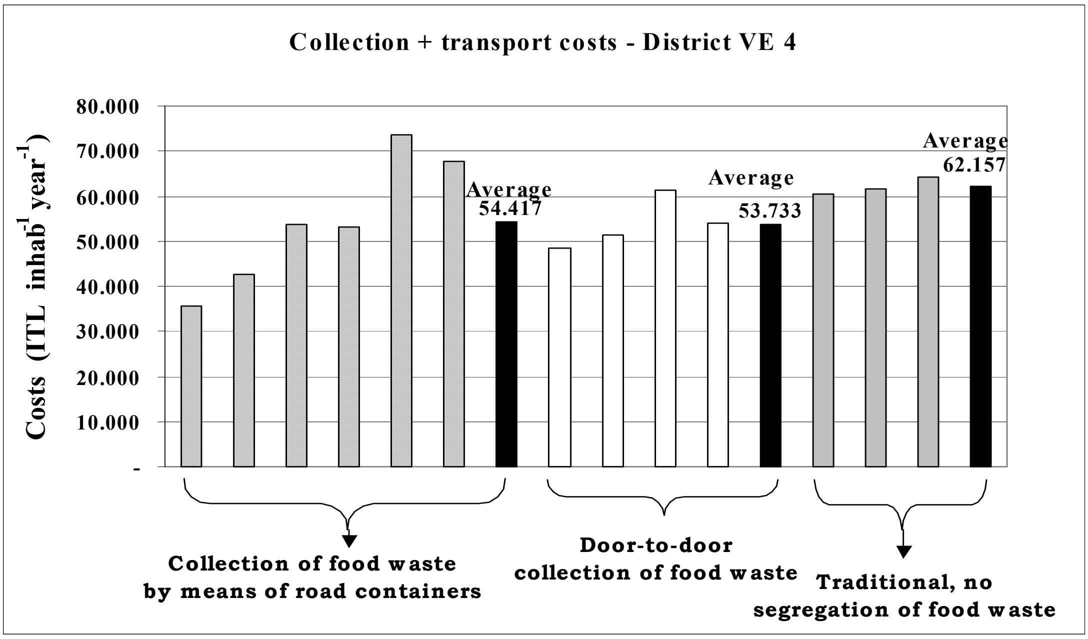
It could be argued that the above discussion is somewhat biased by the inclusion of many examples from Southern Italy in the costs of the road container schemes, as in this instance the climate necessitates more frequent refuse collections. Scuola Agraria del Parco Di Monza carried out a more focused analysis in the district Venezia 4, close to Venice to investigate the relative costs of the different systems in a relatively uniform climate.[53]

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

To understand the unexpected outcomes of the survey, it must be underlined that if source separation of food waste is *added* to that of commingled municipal waste, with no modification in the pre-existing scheme for MSW collection, total costs are likely to rise. This actually tends to happen with the segregation of food waste by means of road containers. But this does not necessarily happen when food waste collection is introduced in such a way that the overall collection system is optimised. The key point is that intensive door-to-door schemes for food waste – when made “comfortable” for households - yield high captures. This sharply reduces the percentage of food waste in residual waste, which can then be collected less frequently with fewer complaints regarding odours. This approach might be considered likely to be especially effective in municipalities where households are charged on the basis of frequency for the collection of residual waste.

[53] Eunomia, *Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis*, Report for WRAP, May 2007

Figure 7-7: Cost Comparison for Different Collection Schemes in a Single District (ITL/inhab/year)



7.4.2.2 Slovak Republic

A study in the Slovak Republic reviewed he costs of existing refuse collection schemes and compared the likely costs of running a weekly food waste collection scheme given the costs of the refuse scheme. The study suggested that a refuse collection system, run on a weekly basis, would cost slightly more than a service which included a weekly food waste collection and a fortnightly refuse collection (see Table 7-6)

Table 7-6: Cost of a team for residual waste collection with rear - loading compactor and food waste collection with open non-compacting vehicle

	N	Cost (SKK/day)	Coll/month (N)	Running cost (SKK/year)	%
Mixed waste					
Driver	1	800	4	38,400	10%
Collector	2	500	4	48,000	13%
Compacting vehicle	1	6087	4	291,186	77%
Open vehicle	0	2475	0	-	0%
Total				**378,586**	**100%**
Integrated Scheme					
Residual waste	**N**	**Cost (SKK/day)**	**Coll/month (N)**	**Running cost (SKK/year)**	**%**
Driver	1	800	2	19,200	10%
Collector	2	500	2	24,000	13%
Compacting vehicle	1	6087	2	146,093	77%
Open vehicle	0	2475	0	-	0%
Total				**189,293**	**100%**
Food waste	**N**	**Cost (SKK/day)**	**Coll/month (N)**	**Running cost (SKK/year)**	**%**
Driver	1	800	4	38,400	21%
Collector	1	500	4	24,000	13%
Compacting vehicle	0	6087		-	0%
Open vehicle	**1**	**2475**	**4**	**118,810**	**66%**
Total				**181,210**	**100%**
Total for Integrated				**370,503**	

Scheme

Source: Scuola Agraria del Parco di Monza (2004) *Draft Handbook for Management of Biowastes: Manual for Slovak Municipalities and Local and Regional Authorities*, http://www.enviro.gov.sk/servlets/files/10432

It concluded:[54]

> *"The integrated scheme (Table 32) seems to be cost effective (the integrated scheme for food waste and refuse is 2% cheaper than traditional refuse collection). The team operating weekly with the open vehicle shows an annual cost (181.200 SKK), which is about 50% the cost that would have arisen if using a compacting vehicle (354.600 SKK)."*

7.4.2.3 Collections of Biowaste

7.4.2.3.1 Investigation of Collection Costs in Germany

A study undertaken in Germany by INFA also investigated the issues affecting the costs of biowaste collection relative to a situation in which biowaste was not separately collected.[55] The report effectively compares

- A collection system of residual waste only (biowaste disposed through this system)

with

- A system which includes a separate collection (i.e. separated from residual waste) for biowaste.

It was assumed that:

- In rural areas:
    - Collection of all waste types is performed with compacting vehicles (side loaders or rear end loaders);
    - Frequencies of collection are lowered when moving from co-mingled collection to residual+biowaste collection to constrain any additional costs;
    - The additional stream of segregated material implies additional investments to facilitate the more diverse collection logistics;
    - The authors explicitly cite the danger of garden waste transfer into the biowaste container. For separate collection of biowaste, typical increases are estimated at 10kg/inhab;
- In urban areas:
    - Collection of all waste types is performed with compacting vehicles (rear end loaders);
    - Frequencies of collection are halved when moving from co-mingled collection to residual+biowaste collection to constrain any additional costs;
    - The additional stream of segregated material implies additional investments to facilitate the more diverse collection logistics; and

[54] Scuola Agraria del Parco di Monza (2004) *Draft Handbook for Management of Biowastes: Manual for Slovak Municipalities and Local and Regional Authorities*, http://www.enviro.gov.sk/servlets/files/10432

[55] INFA (2004) *Cost Consideration of Separate Collection and Treatment of Biowaste* Final Report for Verband der Nordrhein-westfälischen Humus- und Erdenwirtschaft e. V. 19th Nov. 2004

- o The authors claim that specific captures of biowaste (i.e. kg/inhab) in this situation are relatively low.

The key conclusions from the study were as follows:

1. Using the terminology 'rest waste' to mean residual waste, the authors note the following:

   > *According to the investigations, besides the logistic preconditions (vehicle technology, collection frequency etc.) <u>the difference between the disposal cost for the rest waste and the treatment costs of biowaste is decisive</u>. The influence of the mentioned parameters are mentioned in the following figure. Contrary to urban structures with a required biowaste treatment cost difference of at least 55 - 60 €/Mg (biowaste treatment more favourable in costs than treatment of residual waste respectively household waste), in rural structures a difference of approximately 20 - 25 €/Mg is economically sensible.* [our emphasis – see Figure 7-8].

   The key point is that higher captures of biowaste mean the breakeven cost for providing the service is achieved with a lower net difference between the disposal and biowaste treatment costs

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

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

Conversely, savings of approximately 4 €/inh/yr in a rural area and approximately 1 €/inh/yr in an urban area were calculated for two typical disposal areas in Germany (rural / urban) using figures for services and costs usual in the waste sector (see Figure 7-9). This data shows that in the rural situation, where 130 kg/inh/yr of biowaste is collected, total waste management costs reduce by 11%. In the urban situation, where 50kg/inh/yr is collected, total waste management costs reduce by 1%.

Figure 7-8: Difference Between Organic Treatment and Residual Waste Disposal Costs Required for a Cost-neutral Introduction of Organic waste Collection

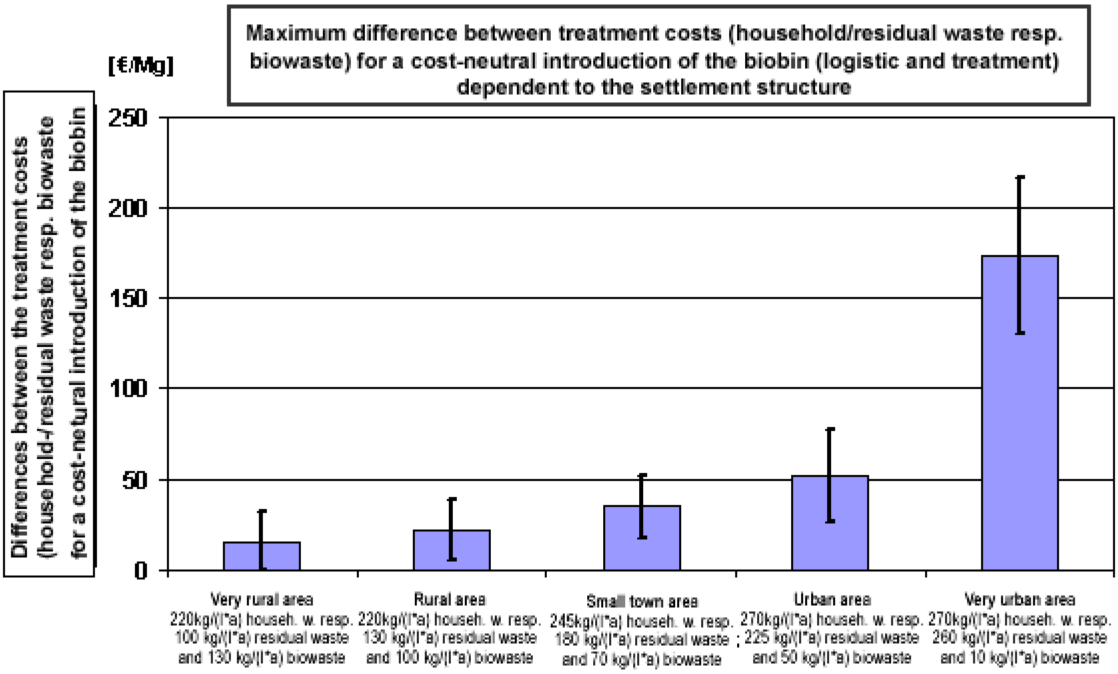
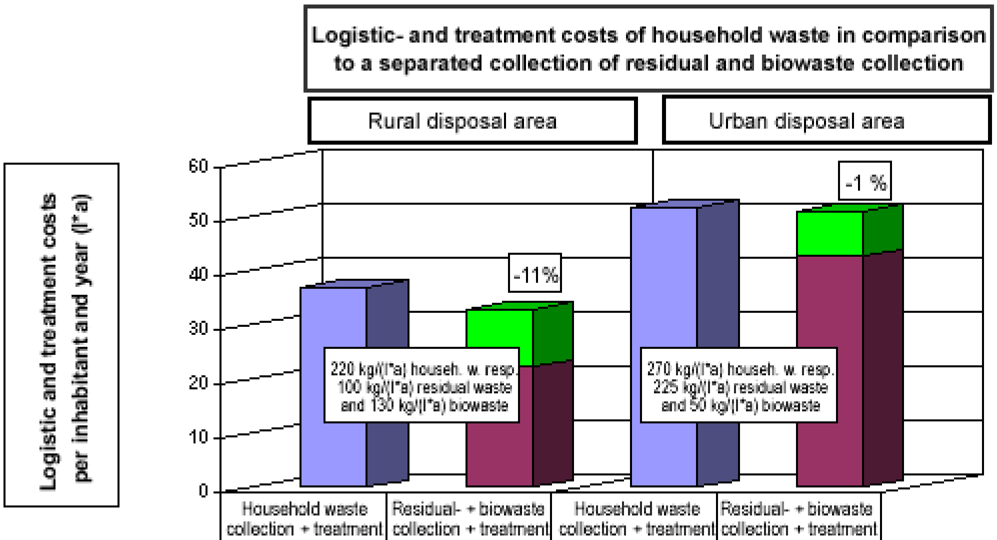


Figure 7-9: Costs for Logistics and Treatment of Segregated Residual and Biowaste Collection Compared to Exclusively Residual (Household) Waste Collection



	Household waste collection + treatment	Residual- + biowaste collection + treatment	Household waste collection + treatment	Residual- + biowaste collection + treatment
Variation as to inhabitant		-11 %		- 1 %
Total costs per inhabitant	36,90 € (I*a)	32,81 € (I*a)	51,61 € (I*a)	50,96 € (I*a)
Total costs per quantity *	168 €/Mg	143 €/Mg	191 €/Mg	185 €/Mg
Biowaste		10,63 €/(I*a)		8,27 €/(I*a)
Residual waste		22,18 €/(I*a)		42,69 €/(I*a)
Household waste	36,90 €/(I*a)		51,61 €/(I*a)	

These results effectively highlight system costs. They are used by the authors to illustrate that, in the overall waste management system, the differences across rural and urban settlement types lead to differing costs of introducing the separate collection system. The suggestion is that the differential between the costs of biowaste treatment and the costs of residual waste treatment needs to be higher in those (urban) areas where the cost of running the separate collection service is deemed to be greater. This is consistent with the general principle that, other things being equal, as the avoided costs of disposal increase, so the financial rationale for separate collection systems becomes stronger. Even so, in this model, only in the very rural area does the differential drop close to zero. This suggests that, perhaps contrary to generally held views, the introduction of the separate collection system is likely to be more beneficial in more rural situations.

The study results to appear to be very strongly conditioned by the capture rate of materials for biowaste treatment. The question of logistics can be decisive in determining the overall financial rationale for separate collection. The method of collection assessed within the INFA study involves large collection vehicles operating the biowaste collection system. The Italian approach, using small non compacting vehicles, would suggest that costs can be further reduced through lower capital outlay on vehicles and associated improvements in fuel efficiency. This would, most likely, reduce the cost differential between biowaste treatment costs and residual waste treatment costs at which the separate collection of biowaste becomes ‘cost neutral’.

7.4.2.3.2 Investigation of Collection Costs in the UK

Detailed cost modelling conducted by Eunomia for WRAP in the UK assessed a wide range of collection scenarios as part of the cost benefit analysis work on options for biowaste.[56] The collection cost model used for this work has been used in around 25% of local authorities in the UK so has been thoroughly benchmarked and optimised to generate representative system costs; as such it has been used in a number of studies to guide national waste and economic policies. A recent study for WRAP was conducted following availability of new data on the operation of small 7.5 tonne (gross vehicle weight) tipping collection vehicles used in trial schemes in the UK.[57] Results taken from this work are shown in Table 7-7. The costs shown are full financial costs (including collection, treatment and disposal) alongside the costs of collection only per household per annum. The systems modelled are shown as net costs compared to a baseline weekly door to door bin-based residual collection service with no separate biowaste collection. Under each scenario, with the introduction of biowaste collection systems, the residual waste collection system is deemed to fall from a weekly basis to a fortnightly door to door bin-based service. In all cases, the movement of waste to and from home composting and centralised recycling centres (civic amenity sites) are accounted for in the mass flows and costs.

As we have noted above, the manner in which the material is collected has implications for the composition of biowaste collected and therefore which treatment systems may be suited. The one system modelled where food and garden waste are collected together is treated by invessel composting. For the other collection scenarios, however, results are shown for the biowaste passing to a range of treatment options.

The data shows that biowaste collection can be conduced at zero or negative economic cost under a number of different situations. Significantly, the results suggest that:

- Where only garden waste is collected separately, the costs are marginally negative (Option A);

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[56] Eunomia (2007) *Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis*, Report for WRAP, May 2007.
[57] A. Gibbs and D. Hogg (2008) *Food Waste Collection: Update to WRAP Biowaste Cost Benefit Study*.

- Where food waste is collected separately on a weekly basis, the collection costs barely change (and the costs can fall if the collection is supported by strong promotion of home composting at households with gardens) (Options C, D, E, F, G, H);
- Where a garden waste collection is added to the service, but with a charge for the garden waste service applied, the costs, net of revenues received, increase marginally (Options J and K);
- If a free garden waste service is included, either with the garden waste material collected in the same container as the food waste (Option I), or alongside the food waste collection (Options L and M), the collection costs increase.

Table 7-7: Total Waste System and Collection Only Costs for Different Approaches to Biowaste Collection Relative to No Biowaste Collection, UK

Switch from baseline with weekly bin based residual collection to fortnightly bin based residual collection and…		**Total System costs per household per annum**	**Collection only costs per household per annum**
A	Fortnightly charged garden waste collection, windrow composting.	-£9	-£8
B	Fortnightly free garden waste collection, windrow composting.	-£2	-£1
C	Weekly food waste collection to digestion. Garden waste collected through civic amenity sites to windrow composting.	-£6	-£0
D	Weekly food waste collection bulked with civic amenity site garden waste to invessel composting.	-£2	-£0
E	Weekly food waste collection bulked with civic amenity site garden waste to digestion.	-£3	-£0
F	Option C plus intensive home composting promotion.	-£8	-£1
G	Option D plus intensive home composting promotion.	-£6	-£1
H	Option E plus intensive home composting promotion.	-£7	-£1
I	Fortnightly combined food and garden waste collection, invessel composting.	£4	£3
J	Option C (digestion) plus charged garden waste collection (windrow composting).	-£2	£1
K	Option D plus charged garden waste collection (invessel composting).	£2	£1
L	Option C (digestion) plus free garden waste collection (windrow composting).	£5	£9
M	Option D plus free garden waste collection (in vessel composting).	£9	£9

The modelling conducted for the results in this table included anaerobic digestion at £50/tonne, invessel composting with controlled mixing of food and garden waste at £45/tonne (scenarios D, G and K), invessel composting with food and garden waste commingled by the householder at £40/tonne (scenario I), windrow composting at £19/tonne, and disposal of residual waste at £71/tonne including tax.

The messages here are important since they highlight the fact that the change in collection costs when separate collection systems for biowaste are introduced depend on the strategy adopted for collecting biowaste. However, they are only significant when two different separate collection services, one for food and one for garden, are both offered free of charge (systems L and M). In the other situations, the cost the separate collection of biowaste is more-or-less offset by the change in refuse collection frequency from weekly to fortnightly.

In particular, they reflect the fact that:

- Introducing weekly collections of food waste only can be as cost-effective as introducing a fortnightly collection of garden waste only. This is due to the fact that the nature of the material being collected can influence the choice of vehicles, and hence, the cost of logistics;
- When introducing collections of both food and garden waste, there are differences in collection cost reflecting whether the materials are collected separately or not, and whether a charge is applied for the collection. Interestingly, the strategy here also affects treatment costs (compare Options C, D and E, which use the same collection strategy, but direct the material to different biowaste treatment options). This is crucial in the context of seeking to secure the additional environmental benefits of anaerobic digestion at an acceptable cost.
- Offering garden waste collections without charging for the collection is likely to increase the quantity of waste collected, increasing the costs of both collection and treatment (compare Options J and M).

It should be noted that no direct changing for residual waste collection occurs in the UK. As such, the incentive for correctly separating biowaste is absent and material capture tends to be weaker than might otherwise be achievable due both to reduced levels of participation and slightly lower capture from those that do participate. There are strong indications that even the best food waste collection systems operating in the UK are currently capturing under 60% of the total food waste arisings. As indicated in Section 7.4.2.3 (and specifically implied in Figure 7-8), higher levels of capture can improve the rationale for and productivity of the collection services, and further improve the balance of costs. There are reasons to believe that such incentives could further enhance the beneficial outcomes from targeted collection of food wastes.

By contrast, where free garden waste collections are operated, the capture of garden waste tends to be extremely high (typically, well above 90%).

7.4.3 Environmental Impacts

Before investigating the environmental impacts associated with transport, it should be appreciated that the relevance of any impacts that may be determined need be considered in the light of the net change in vehicular transport use. The matter of interest for much of this study is not the absolute level of vehicle use, but the likely change in vehicular use when separate collections of biowaste are introduced, and the associated change in environmental damages resulting from this. As we hint at above, vehicular use (whether it be measured in number or weight of vehicles, or total miles driven per annum) may not necessarily increase in the context of biowaste collection. Much depends upon how the system is configured. Indeed, in some systems, because of the changes in vehicles used, fuel use may fall marginally whilst vehicle mileages may increase.

Furthermore, a number of specific externalities associated with vehicular transport are likely, in many countries, to be internalised in fuel and other transport-related duties. If this is the case then the environmental impacts from transport fuel and material use are thereby monetised within the private costs which have already been considered. Some further assessment is, however, necessary to investigate whether this is the case.

The research conducted for the Directorate-General for Energy and Transport for the *“Internalisation Measures and Policies for All external Cost of Transport (IMPACT)”* study gives detailed breakdowns for an extensive range of externality categories.[58] A summary of these categories together with the monetised externalities (shown in euro cents per vehicle kilometre) reproduced from this report are shown in Table 7-8.

Table 7-8: Externalities from Vehicular Transport from The IMPACT Study

External costs for heavy duty vehicles		**€ct/vkm Unit costs (bandwidths)**
Noise	Urban, day	7.01 (7.01 - 17.01)
	Urban, night	12.8 (12.8 - 31)
	Interurban, day	1.1 (0.39 - 1.1)
	Interurban, night	2 (0.72 - 2)
Congestion	Urban, peak	75 (13 - 125)
	Urban, off-peak	0 (-)
	Interurban, peak	35 (0 - 70)
	Interurban, off-peak	0 (-)
Accidents	Urban	10.5 (0 - 13.9)
	Interurban	2.7 (0 - 3.5)
Air pollution	Urban, petrol	(-)
	Urban, diesel	10.6 (10.6 - 23.4)
	Interurban, petrol	(-)
	Interurban, diesel	8.5 (8.5 - 21.4)
Climate change	Urban, petrol	(-)
	Urban, diesel	2.6 (0.7 - 4.7)
	Interurban, petrol	(-)
	Interurban, diesel	2.2 (0.6 - 4)
Up- and downstream processes	Urban, petrol	(-)
	Urban, diesel	3.1 (3.1 - 6.9)
	Interurban, petrol	(-)
	Interurban, diesel	2.7 (2.7 - 6.7)
Nature & landscape	Urban	0 (0 - 0)
	Interurban	1.15 (0 - 1.15)
Soil & water pollution	Urban/Interurban	1.05 (1.05 - 1.05)
Totals		
Urban	Day, peak	109.8 (35.5 - 192)
	Day, off-peak	34.8 (22.5 - 67)
	Night, off-peak	40.6 (28.2 - 80.9)
Interurban	Day, peak	54.4 (13.3 - 109)
	Day, off-peak	19.4 (13.3 - 39)
	Night, off-peak	20.3 (13.6 - 39.9)

Particular impacts listed here, however, are, to a greater or lesser degree, internalised through individual financial mechanisms. Accidents will be covered within insurance costs (included in the financial costs discussed in Section 7.4.2). It is questionable, therefore, whether, or to what degree, accidents should be attributed as external costs.

The more recent IMPACT report makes some specific observations concerning the internalisation of particular impacts:[59]

[58] CE Delft (2008) *Handbook on estimation of external costs in the transport sector (Version 1.1)*, Produced within the study Internalisation Measures and Policies for All External Costs of Transport (IMPACT), European Commission DG TREN, February 2008.

For road transport the main conclusions on existing taxes and charges are listed below.

Fixed taxes and charges *(like circulation taxes and vehicle registration taxes) can give some incentives (e.g. to buy a relatively fuel efficient car) but can not be regarded as internalising external costs of transport activity.*

Kilometre charges *and charges that exceed infrastructure cost levels can be regarded as internalising external costs, particularly when differentiated to relevant parameters (see below). Based on a comparison of existing kilometre based charges, marginal external costs and infrastructure costs we conclude:*

− There are no kilometre related charges on urban and metropolitan roads, while kilometre related costs in metropolitan areas are much higher than in interurban areas, making the gap between charges and costs highest there.

− In many countries there are no kilometre related charges on motorways, therefore also on motorways transport users do not pay their marginal costs. In countries with motorway tolls (either electronically or with office boxes), their level is generally much lower than the marginal costs in congested areas. On non congested motorways, the existing charges do sometimes cover just part of the infrastructure costs, while in a few exceptions they cover or even exceed the total infrastructure and marginal external costs, particularly for small trucks. Note that these conclusions are based on estimates of infrastructure costs which in same cases were calculated with rather rough extrapolations.

− In heavily congested areas, congestion costs are dominant. In non congested metropolitan areas accidents are generally the highest externality, for HGV together with air pollution costs (and for the largest HGV in some countries marginal infrastructure costs as well). For HGV on motorways air pollution and marginal infrastructure costs are the dominant marginal cost components.

Fuel taxes *can internalise fuel consumption related external costs, so particularly climate change costs. Fuel taxes can not be regarded as internalising other external costs because fuel consumption is a very weak proxy for the cost drivers of these external costs.*

This would confirm that fuel related emissions (most specifically climate change and air pollution) are internalised within the environmental costs. Congestion however, remains a potential issue, especially as of the remaining impact categories it is by far the largest.

The significance accorded to (and the approach to valuing) congestion externalities is always a debatable issue, indeed whether these should be considered as ‘external’ costs or not is a matter for debate. It can be argued that a proportion of the costs associated with congestion are not ‘external’ insofar as transport decisions are made on the basis of some knowledge as to when congestion is likely to occur. Indeed, in the waste management case, service providers will be sensitive to congestion-related issues in terms of the timing of their collection rounds (congestion will increase private costs). However, in some situations, it is clear that refuse trucks can add to congestion (especially in narrow streets) so that marginal congestion costs could become quite high, and the assumption that existing duties readily internalise all externalities probably breaks

[59] CE Delft, 2008, *Internalisation measures and policy for the external cost of transport (Version 1.1)*, Produced within the study Internalisation Measures and Policies for All external Cost of Transport (IMPACT), European Commission DG TREN, June 2008.

down.[60] Notwithstanding these points (to the extent that the assumption might not be valid) then to the extent that one is seeking to understand *changes* in the transport externalities across different systems, it can reasonably be argued that collection system changes are unlikely to have a major influence on the analysis. This may be confirmed from the results of the UK collection modelling detailed in Section 7.4.2.3.2 above.

The data in Table 7-9 shows that under the various approaches to biowaste collection (for an authority of 80,000 households) the total number of vehicles does not increase significantly, though where it does the capital cost of the vehicles tends to be lower since smaller vehicles are used (with, arguably, lower associated external costs per tonne of waste collected). Furthermore, the differences in total vehicle mileage driven and fuel costs (obviously linked to fuel use) either decrease under the biowaste collection options or do not increase significantly.

Table 7-9: Collection Vehicle Specific Indicators from UK Biowaste Collection Modelling

Collection system		**Total number of collection vehicles**	**Capital cost of vehicles**	**All vehicle miles driven / annum**	**Total fuel usage, litres of diesel per annum, all vehicles**
-	Baseline (weekly bin based residual)	20	£2,270,242	252,980	255,571
A	Fortnightly charged garden waste collection, windrow composting.	17	£1,938,874	223,923	213,434
B	Fortnightly free garden waste collection, windrow composting.	18	£2,083,905	225,624	227,935
C	Weekly food waste collection to digestion. Garden waste collected through civic amenity sites to windrow composting.	23	£1,842,478	323,773	207,105
D	Weekly food waste collection bulked with civic amenity site garden waste to invessel composting.	23	£1,842,478	323,773	207,105
E	Weekly food waste collection bulked with civic amenity site garden waste to digestion.	23	£1,842,478	323,773	207,105
F	Option C plus intensive home composting promotion.	23	£1,824,762	322,760	206,081
G	Option D plus intensive home composting promotion.	23	£1,824,762	322,760	206,081
H	Option E plus intensive home composting promotion.	23	£1,824,762	322,760	206,081
I	Fortnightly combined food and garden waste collection, invessel	20	£2,288,357	260,703	263,373

[60] For example, work for the then DETR looks at the marginal costs of congestion using linear (with respect to vehicle quantities) speed-flow curves, and assuming that the value of time should be valued at something close to the average wage rate (see T. Sansom, C. Nash, P. Mackie, J. Shires and P. Watkiss (2001) *Surface Transport Costs and Charges: Great Britain 1998*, Report for DETR, July 2001). All these (and other) assumptions are open to question to a degree. This approach tends to give high marginal costs for congestion. It gives highest costs for dense urban areas at peak hours, arguably the very time when most people are likely to *internalise* these costs into their decision-making processes. From this point of view, it is questionable whether the attribution of congestion costs as a 'marginal externality' is necessarily correct, albeit it is an accepted approach in the literature.

	composting.				
J	Option C (digestion) plus charged garden waste collection (windrow composting).	27	£2,207,978	373,118	244,172
K	Option D plus charged garden waste collection (invessel composting).	27	£2,207,978	373,118	244,172
L	Option C (digestion) plus free garden waste collection (windrow composting).	28	£2,404,380	377,640	261,523
M	Option D plus free garden waste collection (invessel composting).	28	£2,404,380	377,640	261,523

It is possible to calculate anticipated externalities from the data from Table 7-8 and Table 7-9. However, it would not be right to use the €/km externalities from the IMPACT study (Table 7-8) for all the impact categories against the distances in Table 7-9. Air pollution and climate change are related to fuel usage and not distances travelled. The same is true for 'up- and downstream processes' (costs arising from energy and fuel production) and soil and water pollution (the most significant effects of traffic on soil come from the emission of heavy metals and polycyclic aromatic hydrocarbons). We may however choose to use the per kilometre IMPACT externalities for the impacts to nature and landscape since these essentially relate to (road) infrastructure and not its use. The IMPACT report admits that the nature and landscape marginal costs are very low. Noise impact may also be assumed to be more closely correlated to the distances travelled than they are the fuel use.

We are able to summarise the likely externalities of biowaste collection relative to no biowaste collection using the data in the tables above. We calculate externalities per household relating to noise, nature and landscape according to the relative travel distances between the biowaste collection options and the baseline. We then calculate the total CO_2 emissions associated with fuel use and use a €27 / tonne unit damage cost to generate relative damages per household compared to the non biowaste baseline.[61] Other fuel related damage costs are calculated on a pro rata basis to the CO_2 damages according to the central urban unit cost externalities from the IMPACT study in Table 7-8. Accidents and congestion are not included for the reasons discussed above. According to these methodologies, Table 7-10 summarises the total externalities associated with the introduction of separate biowaste collection, for the various collection systems considered.

Table 7-10: Collection Vehicle Specific Indicators from UK Biowaste Collection Modelling

Collection system		Relative noise, nature & landscape damage costs per household	Total CO_2 from combustion of fuel, tonnes	Relative CO_2 damage costs per household	Correlated other fuel related damage costs per household	Net external damage costs relative to no biowaste collection
-	Baseline (weekly bin based residual)	-	573	-	-	-
A	Fortnightly charged garden waste collection, windrow composting.	-0.05 €	478	-0.03 €	-0.06 €	-0.11 €
B	Fortnightly free garden waste collection, windrow composting.	-0.04 €	511	-0.02 €	-0.04 €	-0.08 €

[61] The choice of damage costs is set out in Annex F.

C	Weekly food waste collection to digestion. Garden waste collected through civic amenity sites to windrow composting.	0.12 €	464	-0.03 €	-0.07 €	0.05 €
D	Weekly food waste collection bulked with civic amenity site garden waste to invessel composting.	0.12 €	464	-0.03 €	-0.07 €	0.05 €
E	Weekly food waste collection bulked with civic amenity site garden waste to digestion.	0.12 €	464	-0.03 €	-0.07 €	0.05 €
F	Option C plus intensive home composting promotion.	0.11 €	462	-0.03 €	-0.07 €	0.04 €
G	Option D plus intensive home composting promotion.	0.11 €	462	-0.03 €	-0.07 €	0.04 €
H	Option E plus intensive home composting promotion.	0.11 €	462	-0.03 €	-0.07 €	0.04 €
I	Fortnightly combined food and garden waste collection, invessel composting.	0.01 €	590	0.01 €	0.01 €	0.02 €
J	Option C (digestion) plus charged garden waste collection (windrow composting).	0.20 €	547	-0.01 €	-0.02 €	0.18 €
K	Option D plus charged garden waste collection (invessel composting).	0.20 €	547	-0.01 €	-0.02 €	0.18 €
L	Option C (digestion) plus free garden waste collection (windrow composting).	0.20 €	586	0.00 €	0.01 €	0.21 €
M	Option D plus free garden waste collection (invessel composting).	0.20 €	586	0.00 €	0.01 €	0.21 €

We can see from the table that in all cases, the externalities are a scale of magnitude smaller than the relative financial cost differences between collection systems in Section 7.4.2. As such, in light of the fact that specific approaches to collection can reduce net financial collection costs, we may well assume that any external costs that are created in the context of a switch to biowaste collection are accommodated in the potential reductions in financial costs. In any case, the levies placed on fuel and other transport related taxes are likely to internalise much (if not all) of the calculated external costs here.

7.4.4 The Issue of Household Time

Some cost-benefit studies related to recycling systems – among them, those of Radetzki, Bruvoll and Sterner and Bartelings[62] – seek to impute a cost associated with time involved in waste-related activity in the household. Where this does occur, then to the extent that one seeks to understand the costs and benefits of a charging scheme, it should, of course, be the case that a charging scheme is only implicated in the 'incremental cost' of the time and resources spent by the householder above and beyond what was already happening in the absence of charging.

7.4.4.1 On time inputs and convenience

Radetzki imputes a cost calculated at the hourly rate paid to untaxed manpower for household services at around $7.5 per hour (in 2000), equivalent to approximately €8.40 per hour in 2005. Even if one assumes such costs should be identified, this level might be questioned. Markandya, for example, valued non-working time at 15% of the gross wage

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62. M. Radetzki (2000) *Fashions in the Treatment of Packaging Waste: An Economic Analysis of the Swedish Producer Responsibility Legislation*, Brentwood: Multi Science; A. Bruvoll (1998) Taxing Virgin Materials: An Approach to Waste Problems, Resources, *Conservation and Recycling*, 22, pp.15-29; and T. Sterner and H. Bartelings (1999) Household Waste Management in a Swedish Municipality: Determinants of Waste Disposal, Recycling and Composting, *Environmental and Resource Economics*, 13, pp. 473-91.

rate (though the basis for the figure is not made clear in the context).[63] Bartelings uses these other studies as a basis for making assumptions concerning the costs of separation of organic wastes.[64]

Relatively little attention is given by any of these authors as to the effect of the nature of the service provided to householders on the time spent dealing with waste. The estimates of time input from studies such as those by Bruvoll *et al.* do not sit easily alongside much of the discussion of how to optimise service provision for the simple reason that the systems are (by inference) incredibly inconvenient (the study includes 9 hrs per year transport to collection points), and whilst this might imply an inconvenient scheme for dry recyclables, it would be extremely unlikely that such a system could be used for the collection of food waste.[65] Where the collection of recyclables is exclusively through bring schemes, a general cost-benefit analysis ought to account for the fact that the unit costs of recycling for the municipality would (or should) be correspondingly low since much of the increase in costs will be borne by the householders themselves.

Generally, one might consider, therefore, two ends of a spectrum for municipalities implementing recycling schemes:

- Those which implement inconvenient recycling systems which impose time costs, and for that reason, will tend to deliver relatively low captures of material; or
- Those which implement convenient, quality services which impose little or no additional time cost (the household simply uses the right container) and deliver higher captures of recyclable material. These services, at least from the perspective of collection alone, are, usually, more expensive at the point of delivery. However, the net system costs are also heavily influenced by the costs of disposal being avoided, so where collection systems are well-designed, increased collection of organic wastes can lead to net savings owing to reduction in the costs of collection of refuse.

In other words, if one is considering *convenient*, high-quality kerbside schemes, making use of estimates for time cost derived from studies focusing on *less convenient* bring systems, and also, those focused on dry recyclables, is likely to be a mistake.[66]

This is the first reason why focusing heavily on time in the context of DVR charging schemes might not be relevant, unless the scheme is inconvenient, in which case, issues of dumping will probably also arise. It is generally recommended that charging systems, because of the potential incentive they generate to dump illegally (the significance of which is contested in the literature, apparently less so in practice), should be introduced only where good quality recycling infrastructure is already in place.

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[63] Anil Markandya (1998), *The Indirect Costs and Benefits of Greenhouse Gas Limitation*, Report prepared for the UNCCEE, Roskilde, DK, cited in RPA (Risk and Policy Analysts) and Metroeconomica (1999) *Induced and Opportunity Cost and Benefit Patterns in the Context of Cost-Benefit Analysis in the Field of Environment, Final Report to European Commission*, DG III, February 1999.

[64] H. Bartelings (2003) *Municipal Solid Waste Management Problems, An Applied General Equilibrium Analysis*, PhD Thesis, University of Wageningen.

[65] See A. Bruvoll, B. Halvorsen and K. Nyborg (2000) Household Sorting of Waste at Source, *Economic Survey* 4/2000, pp.26-35 and A. Bruvoll, B. Halvorsen and K. Nyborg (2002), Households' Recycling Efforts, Resources, *Conservation and Recycling*, 36: 337-354. It is a feature of some Scandinavian collection systems that greater reliance is placed upon bring recycling centres than on kerbside collection of recyclable materials. This could be compared with, for example, Germany, Austria, Belgium, Italy, United Kingdom, or Australia, etc. where rather greater emphasis is placed upon doorstep recycling. Time cost issues are likely to be more an issue in the former than the latter group of countries / set of circumstances.

[66] It is, in our view, surprising that studies which discuss time costs do not spend more time describing the available waste management infrastructure in more detail, as well as the possible alternatives.

7.4.4.2 Potential Private Gains in Utility

Regarding household costs, Porter takes the view that the benefits which people may derive from participating in recycling are likely to be roughly equal to the social costs of engaging in the activity.[67] He makes the point that some detractors of recycling have derived large costs for this activity. On the other hand, he adds that a positive willingness to pay for recycling is often identified in the literature.[68]

For Smith, the issue as to whether or not additional time spent in recycling should be included in an analysis of costs and benefits of extended producer responsibility turns on whether the engagement with the activity is voluntary or enforced:[69]

However, household time and household direct expenditures on cleaning, sorting and transporting waste products should not be included in an assessment of the overall costs and benefits of EPR [extended producer responsibility], where households undertake these actions on a voluntary basis. Where household costs are incurred voluntarily, the inference might be drawn that the household experiences counterpart benefits, in the form of satisfaction – or a "warm glow" – from their environmentally-responsible behaviour, that are at least as large as any costs incurred. If this view is taken, then the costs incurred voluntarily by households should be omitted, so long as the "warm glow" benefits too are omitted. The implication is that, in the case of an EPR programme where households voluntarily choose to participate (and where they can, instead, choose to discard their waste in other ways), there is no need to include any estimate of household costs in the cost-benefit analysis of the programme.

On the other hand, there is a case for including at least some measure of household costs, where households do not incur these costs voluntarily. Where households are compelled by law to separate their wastes, or are required to transport their wastes to inconveniently-located collection facilities, some, at least, may perceive this as an onerous task, from which they gain no corresponding "warm glow". Others may be happy to do this without compulsion, and may perceive no cost. It is then a matter, in principle, for research to determine what proportion of the population perceive the programme as imposing onerous requirements, and how large the perceived costs of the programme are to these individuals. The only household costs that should be included in the analysis are those borne by households who would not act in the absence of compulsion.

Following this line of argument, then wherever schemes do not *compel* households to segregate materials, one might be inclined to the view that time devoted to the activity should not be included as a cost.

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[67] Richard C. Porter (2005), Benefit-cost Analysis and the Waste Hierarchy – US Experiences, in Environmental Assessment Institute (2005) *Rethinking the Waste Hierarchy*, EAI: Copenhagen.

[68] For example, Jakus *et al.* carried out studies to elicit willingness to pay for recycling, and estimated this at £5.78 per household per month, whilst Tiller *et al.* report that in Tennessee, households would pay $4 per month (on the basis of contingent valuation) (see P. M. Jakus et al. (1996) Generation of Recyclables by Rural Households, *Journal of Agricultural and Resource Economics*, Vol 21 (1), pp 96-108; K. H. Tiller et al. (1997) Household Willingness to Pay for Dropoff Recycling, *Journal of Agricultural and Resource Economics*, Vol 22 (2), pp 310-320). In seeking to understand some measure of the value of household time used in separation activity, Bruvoll, Halvorsen and Nyborg report that people in Norway on average would be willing to pay a significant amount for *others* to do the recycling as long as the same environmental benefits result (A. Bruvoll, B. Halvorsen and K. Nyborg (2002), Households' Recycling Efforts, Resources, *Conservation and Recycling*, 36: 337-354). The study finds that some households would not wish to see this happen even when it is free (suggesting they themselves gain some benefit from the activity) whilst others would be willing to pay for the activity to occur even though it was offered at zero cost. This is interpreted by the authors as a basis for estimating the cost of householders' time but other interpretations clearly exist, not least of which is that this is a measure of the value placed by householders on the activity of 'recycling', irrespective of who it is done by. Why else would households be prepared to pay others to carry out the activity for them?

[69] S. Smith (2005) *Analytical Framework for Evaluating Costs and Benefits of EPR Programmes*, Report for OECD Working Group on Waste Prevention and Recycling, ENV/EPOC/WGWPR(2005)6/FINAL, http://appli1.oecd.org/olis/2005doc.nsf/linkto/env-epoc-wgwpr(2005)6-final.

On the other hand, there may be incentives under charging schemes which act so as to alter the level of effort, and associated time input, related to separation. Consequently, the way in which additional segregation / waste reduction efforts associated with such systems should be treated is probably through seeking to:

1. identify the incremental change in costs of time devoted to the activity;
2. identify the utility derived by those engaging in the activity; and
3. quantify the net effect in terms of cost and benefit.

It is a somewhat open question as to what the net outcome might be. Such an analysis is far from straightforward to carry out. In cases where time costs are included, there is clearly a danger that no counterpart benefit is acknowledged, and this appears to have been the case. Many households opt-in to recycling schemes without the need for any financial incentive. Presumably, therefore, where citizens alter their behaviour in response to charging systems, they do so quite wittingly in the context of their own cost-benefit calculus.[70]

It remains the case that the nature of service provision can do much to reduce the effort required by householders, and reduce the requirement for space in households where this is a constraint (for example, by raising collection frequencies and providing appropriate means of containment). Cost-effective waste management is not simply about using incentives to alter household behaviour. It is also about careful design of collection systems to ensure efficient capture of quality materials. Evidently, the private costs of the waste management system must remain a consideration for those with the responsibility for providing such a service.

7.4.4.3 Establishing Norms of Behaviour

From a more institutionally informed perspective, one might argue that charging systems seek to establish, or strengthen, a norm of behaviour in which materials which can be recycled are recycled using the services provided. In the extreme, some municipalities have sought to effectively enshrine a rights structure which makes it a duty for citizens to segregate some materials. Through changing the rights structure, what is defined as the acceptable norm is transformed. Elsewhere, such formal sanctions may not be necessary as norms of behaviour change, in which case, the same effect can occur through the medium of informal institutional changes. In some studies, households have thought they were operating under a mechanism of compulsion when in fact, they were not.[71]

Under either circumstance, the fact that separating wastes becomes, either formally or informally, a duty (dependent upon the rights structure) makes it somewhat awkward to impute a labour cost element for the activity. At the same time, those designing recycling schemes (or for that matter any scheme which seeks to elicit public participation, for example, responsible handling of litter) must make the process easy for the public to participate in. The challenge then is to minimise the financial costs of service provision, consistent with providing a service of the desired quality and scope, with the service seeking to induce sustainable waste management behaviours.

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[70] One study has even suggested that introducing charging systems can dissuade *some* households from recycling as they 're-frame' their decisions. In other words, some households may switch between voluntaristic pro-social behaviour to a more 'hard-headed' calculus of costs and benefits in the wake of the introduction of charging schemes, leading them to reduce (or sometimes stop) recycling activity (see Thorgersen (1994)).

[71] A. Bruvoll, B. Halvorsen and K. Nyborg (2000) Household Sorting of Waste at Source, *Economic Survey* 4/2000, pp.26-35 and Eunomia (2004), *Compulsory Recycling Scheme Review and Second Phase Roll Out Plan*, Final Report to the London Borough of Barnet.

7.4.5 Net Costs

We have determined above that the difference in financial costs from collection of biowaste can, if the transition in collection systems is well engineered, be a net reduction in collection costs.

In terms of the external costs, the issue relates to the impacts over and above those associated with vehicular use *without* biowaste collection. Again we show that it is possible to re-optimise services in the transition to biowaste collection such that there is no significant increase in transport related externalities. The costs that may arise are likely to be a scale of magnitude smaller than the associated financial costs. Furthermore, what impacts may arise may also be said to have already been internalised through fuel and other vehicle duties.

The costs of particular services can be variable depending on methods of collection. However, we have shown that biowaste collection (in the context of integrated collection systems) can be undertaken with zero additional cost. As such, we do not value the contribution from collection costs and impacts within the results in the modelling through the remainder of this report.

7.4.6 Practicalities of Achieving Improved Biowaste Capture for Individual Member States

The degree to which targets for biowaste recycling will necessitate reform of collection systems depends upon both the level of the target and the manner in which it is specified. Furthermore, the pre-existing collection infrastructure will have implications for what will be required to improve performance.

<u>Collection System Considerations</u>

Policies demanding increased rates of biowaste recycling will be expected to require changes in the approach to dealing with biowaste. We may consider the measures which can help to deliver improvements in biowaste capture as falling into three broad categories:

- **Optimisation of existing systems**:
    - **Collection frequency adjustments**. As we discuss above, provision of a more frequent organic waste collection frequency, coupled with a reduced residual waste collection frequency, provides a more positive stimulus to use the service effectively.
    - **Charging system adjustments**. Altering the structure of charges in respect of segregated organics and disposal can generate price responses, and influence overall diversion of material from disposal. There is merit in levying a charge on the organic waste collection as this maintains an incentive for waste prevention and home composting, as well as ensuring that contamination of the biowaste bin is not made problematic by virtue of having a zero marginal cost;
    - **Other promotion / incentivisation measures**. Together with providing households with the information on service operation, further examples of promotional activity may include door knocking campaigns through to lottery style campaigns or school reward schemes etc. Such promotional campaigns can generate uplifts in performance, whilst poorly communicated collection systems may be expected to underperform.

- **Adaptation of existing systems:**
    - **Changes in materials collected**. Wheeled bin collections of biowaste, in particular, are used across Europe for a number of different materials – principally garden waste, food waste and card. In many situations, only garden waste is collected. Here, the addition of food waste to the collection is a relatively simple step for the collection system; however, as discussed above this approach will significantly increase treatment costs and tends to result in poor capture of kitchen waste. By contrast, in some situations, small containers (30 litres and sometimes less) are used to collect food waste only. Other things being equal, where food waste is not collected, captures may be lower. In areas with gardens, where garden waste is generated, arguably, the potential for home composting exists.

      The addition of garden waste segregation at civic amenity sites / containerparks also allows more material to be captured for recovery.
    - **Increased scheme coverage**. Providing collection services to additional households in an authority with an existing scheme will generally increase captures, Certain households, of course, may not be suitable for particular collection systems – the most obvious example being properties without gardens having no facilitation for a green waste collection service.
- **Replacement of existing, or introduction of new, collection systems**
    - **New schemes**. The introduction of new schemes typically involves the planning of new systems, their design, and possibly (depending upon who is to be tasked with the service) a new tender for a new collection service. This planning phase may incur up front costs. These costs will, on a per household basis, generally depend upon the size of municipality / municipalities. We expect these studies to cost of the order €1 per household for consideration of the system, but this would be a one-off cost. New services also need to be communicated to residents. Typically, municipalities communicate regarding whatever collection system they have. In principle, therefore, the costs of communication should be ongoing, irrespective of system design. However, new services generally require additional communication effort in the preparatory phase for the change. Once again, these might be of the order €1-2 per household over and above ongoing communications efforts. It should be noted that these costs, as well as any additional upfront costs in seeking to plan for biowaste treatment facilities, may or may not offset requirements for studies in advance of new residual waste treatment facilities. Typically, the magnitude of the investments in residual waste treatment can lead to significant expenditure in the preparatory phases. However, it is difficult to know in the policy scenarios modelled below whether changes in demand for residual waste treatment would occur through changes in numbers of facilities, or reduced scale at the same number of facilities that would otherwise have been required.

Impact of Level of the Target

Observation of current performance across individual member states suggests that there are a number of countries which are recycling garden waste relatively effectively but food waste tends to be somewhat less well captured. No country captures food waste as effectively as some countries capture garden waste (the latter is claimed at greater than 90% in some countries). A situation which is fairly common is where the system collects food and garden waste together, often conducted through a bin based approach. In many such systems operating around Europe today it is likely that the garden waste capture meets the 90% objectives of Scenario 2, but food waste capture may be less than the 60% target.

Deployment of collection systems targeting garden waste only is observed in countries where kerbside collections of biowaste are relatively absent. In such systems, which may rely on civic amenity sites, or containerparks, there may be effective diversion of garden waste from the residual stream. However, because the food waste element may typically be expected to be considerable, especially in urban areas, overall biowaste capture tends to be much lower than it could otherwise be.

There is growing evidence to suggest that highest captures of biowaste are achieved where food and garden waste are targeted independently, and the collection systems are configured in a manner that is suited to the individual materials. Being the more readily degradable and odorous material, food waste typically requires more regular collection than garden waste. Evidence suggests that regular collection of food waste combined with a less frequent garden waste collection service (or well provisioned network of recycling centres for this material) may lead to the best outcomes.

The issue of how countries, and the individual local authorities who provide the collection services, meet recycling targets depends very much on the level of that target and which systems are assumed to be in place in future. Targets demanding a simplistic overall biowaste recycling rate of, for instance, 50% may be achievable by simple incremental adaptation of existing garden waste collection systems to incorporate food waste (or provision of this service from new). Higher targets (particularly if specified for the food and garden waste independently) are more likely to require a move to independent collection systems.

In Scenario 3 (see Chapter 10), many countries are meeting, or close to meeting, the required target in the baseline. It seems quite possible that, depending upon how countries implement schemes in future, the targets can be met by simple adaptation of existing schemes. Only a small number of countries would necessarily require significant implementation of new collection services.

In Scenario 2 (see Chapter 9), the likely requirement is that in some countries, additional effort would be required in respect of garden waste captures, but the effort required to capture additional food waste will be required of rather more countries. Rather more countries would be expected to need to implement new collection services. The nature of this requirement depends, as discussed above, upon how countries meet the levels of performance they are expected to achieve in the baseline. As discussed above, there are likely to be some set up costs, but these are likely to be relatively small, of a one-off nature, and possibly offset by savings in respect of planning for alternative (residual waste) treatment.

7.5 Composting

7.5.1 Financial Costs

Table 7-11 examines the financial costs under the social cost metric for an in-vessel or enclosed composting facility. These costs vary, being lowest in the lower labour cost countries.

We have assumed that the costs of in-vessel composting of food waste are effectively higher than for a tonne of mixed biowaste by virtue of the fact that garden waste is treatable at lower cost windrow facilities.

7.5.2 Environmental Impacts

Within the sections on the environmental impacts of the different treatment options that follow, the external damage costs associated with the different options are outlined in detail using the results obtained within our analysis for the Czech Republic.

The Czech Republic relies on coal and nuclear for its electricity generation. Whilst the latter accounts for 30% of the total, the former accounts for nearly 60%. The country also has a heavy dependence on coal and oil for its heat generation, which together account for approximately 50% of the total generation mix. As a result, the damage costs associated with electricity and heat production are higher than the average taken across all EU member states.[72]

The external costs associated with air pollution are close to the EU average for most pollutants although the damages associated with NOx pollution are lower than the average.

Table 7-12 presents the results for the Czech Republic for in-vessel composting facilities. The table identifies the impacts associated with the process itself (including the use of energy at the facility) and those emissions avoided by the use of the compost produced by the plant. Whilst the climate change and air quality impacts relate to emissions to air, those associated with the use of compost include impacts to soil and water.

The damage costs relate to the following environmental impacts resulting from the in-vessel composting process:

- Direct emissions from the process are principally biogenic CO_2 emissions from the degradation of food during the composting process.
- Air quality impacts resulting from the process itself (excluding the energy impacts) are relatively small, and relate to NH_3 emissions.
- Impacts associated with energy used by the process are dominated by the damage caused by air quality emissions. A significant proportion of this impact results from the use of electricity. Principal emissions are SOx from the combustion of coal, along with some emissions of NOx from both diesel and electricity use.[73]

[72] The average carbon intensity for electricity generation across the EU in 2006 was 430 g of CO_2 equivalent per kWh, whilst the comparable figure for heat was 270 g CO_2 equivalent. Results for the Czech Republic were 554 and 320 g CO_2 equivalent respectively.

[73] These impacts are reduced for countries that generate a significant proportion of their electricity from nuclear, gas, hydro-power or wind.

Table 7-11: Financial Costs for IVC (Social Metric)

Country	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI	FR	DE	EL
Unit Capex	€205.1	€218.1	€218.1	€218.1	€157.0	€171.2	€167.6	€220.7	€163.5	€209.0	€215.1	€209.1	€171.2
Discount Rate	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%
Annualised Capex	€15.09	€16.05	€16.05	€16.05	€11.55	€12.60	€12.33	€16.24	€12.03	€15.38	€15.83	€15.39	€12.60
Unit Opex	€12.5	€13.0	€13.0	€13.0	€10.7	€11.3	€11.1	€13.1	€11.0	€12.6	€12.9	€12.7	€11.3
Maintenance	€10.26	€10.91	€10.91	€10.91	€7.85	€8.56	€8.38	€11.04	€8.18	€10.45	€10.75	€10.46	€8.56
Total (NPV per Tonne)	€38.3	€40.6	€40.6	€40.6	€30.0	€32.4	€31.8	€41.0	€31.1	€39.0	€40.0	€39.0	€32.4

Country	HU	IE	IT	LV	LT	LU	MT	NL	PL	PT	RO	SK	SI	ES	SE	UK
Unit Capex	€166.6	€205.0	€205.1	€159.6	€161.2	€218.9	€171.2	€211.2	€165.4	€176.0	€158.6	€163.8	€176.3	€185.6	€209.0	€205.0
Discount Rate	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%
Annualised Capex	€12.26	€15.08	€15.09	€11.74	€11.86	€16.11	€12.60	€15.54	€12.17	€12.95	€11.67	€12.05	€12.97	€13.66	€15.38	€15.08
Unit Opex	€11.1	€12.5	€12.5	€10.8	€10.9	€13.0	€11.3	€12.7	€11.1	€11.4	€10.8	€11.0	€11.4	€11.8	€12.6	€12.5
Maintenance	€8.33	€10.25	€10.26	€7.98	€8.06	€10.94	€8.56	€10.56	€8.27	€8.80	€7.93	€8.19	€8.81	€9.28	€10.45	€10.25
Total (NPV per Tonne)	€31.6	€38.3	€38.3	€30.4	€30.7	€40.7	€32.4	€39.3	€31.4	€33.3	€30.3	€31.2	€33.3	€34.9	€39.0	€38.3

Table 7-12: Indicative External Damage Costs for In-Vessel Composting

	Climate change	Air quality	Other impacts	Totals
PROCESS				
Direct emissions (non energy)	€9.94	€0.49		€10.57
Energy use (electricity & diesel)	€0.44	€4.57		€5.11
USE OF COMPOST				
CO_2 emissions from soil	€1.87			€1.87
Diesel used to spread compost	€0.02	€0.58		€0.60
Reduction in pesticide use			- €3.29	- €3.29
Nutrient displacement impacts			- €1.96	- €1.96
Avoided energy, fertiliser production	- €0.09	- €0.25		- €0.34
Avoided phosphate rock extraction			- €0.69	- €0.69
Avoided water use			- €0.82	- €0.82
Avoided nitrogen leaching			- €1.61	- €1.61
Avoided N_2O emissions			- €0.13	- €0.13
Avoided peat extraction			- €0.65	- €0.65
FINAL TOTALS	€12.18	€5.39	- €9.15	€8.66

- Although no emissions are offset by energy generation, the avoided emissions associated with the use of compost offset a considerable proportion of the total emissions from the process. The most significant impacts here are associated with the reduced use of nitrogenous fertiliser.

Section 7.5.2.1 confirms the impacts that are included within the damage costs presented in Table 7-12, whilst Section 7.5.2.2 identifies impacts that have not been included within the monetised damages presented above. Further information is provided in Annex F.

7.5.2.1 Monetised Environmental Impacts

Our modelling of the environmental impacts of IVC composting assumes a well managed process designed to minimise CH_4 emissions. We assume the facility uses a biofilter to reduce emissions of NH_3, VOC and CH_4 but that this occurs at the expense of some additional N_2O emissions. The plant is assumed to use 40 kWh of electricity per tonne of waste treated at the facility. A small amount of diesel (0.3 litres) is used during the composting process, and a similar amount is later used to apply the compost to the soil. Our model includes impacts associated with both greenhouse gas emissions as well as emissions to air of the other air pollutants required during the generation of all energy sources.

We attribute external costs to emissions of greenhouse gases and other pollutants to air, including both the direct emissions to air (from the composting process) and the indirect emissions (associated with energy use by the process).[74]

Impacts are calculated on the basis of one tonne of biowaste to the composting process. The model accounts for and monetises all CO_2 emissions, including those generated from

[74] AEA Technology (2005) Damages per Tonne Emission of $PM_{2.5}$, NH_3, SO_2, NOx, and VOCs for each EU25 Member State (Excluding Cyprus) and Surrounding Seas, Report for CAFÉ Programme, March 2005

the biogenic carbon contained within biowaste which are typically ignored when a life-cycle analysis approach is taken.

We include within our model the slow release of CO_2 from the soil after the compost is added. After 50 years, approximately 13% of the initial carbon contained within the compost remains in the soil.

Our model assumes 400 kg of compost is produced per tonne of waste to the facility. 50% of the compost produced is assumed to be used in agriculture. We consider the following benefits associated with the use of compost in this way:[75]

- The displacement of alternative nutrient sources otherwise applied through the use of synthetic fertiliser, including the avoided external costs of fertiliser manufacture and the avoided energy use associated with this.
- The greenhouse gases avoided from nitrogenous fertiliser applications (i.e. N_2O emissions) and the external costs associated with this.
- Avoided external costs from a reduction in the leaching of nitrate (from nitrogenous fertilisers) into groundwater.
- Avoided external costs associated with process wastewater and phosphogypsum disposal during the manufacture of phosphate fertiliser.
- Avoided energy requirement associated with the mining of phosphate rock for phosphate fertiliser, and the avoided external costs associated with this
- Avoided external costs through a reduction in the use of pesticides
- Avoided external costs through a reduction in the use of water.

The remaining 50% of the compost is assumed to displace the use of peat in horticulture and hobby gardening applications. Here the avoided impacts are principally the slow release of CO_2 from the aerobic degradation of peat after its removal from the peat-land.[76]

7.5.2.2 Environmental Impacts which are not Monetised

We have not quantified the external *costs* associated with any of the following:

- The production of leachate from composting;
- Odour from composting process, and other nuisances such as flies and vermin;
- External costs associated with the production of bioaerosols;
- The impacts associated with human or plant pathogens; and
- Estimation of the disamenity associated with a composting facility.

We note that the majority of the above can be minimised with careful plant maintenance and process management.

The build up of metals in soil is a recognised issue; however, composts from source segregated materials appear to give rise to limited impacts in this regard as long as applications of compost are in line with good agricultural practice.[77] Indeed, the thematic

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[75] For a detailed description of the methodology used to calculate these estimates see: Eunomia (2007) *Managing Biowastes from Households in the UK: Applying Life-cycle Thinking in the Framework of Cost-benefit Analysis*, Appendices to the Main Report, Report for WRAP, May 2007

[76] This follows the methodology described in AEA Technology (2001) *Waste Management Options and Climate Change: Final Report*, European Commission: DG Environment, July 2001

[77] See F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

strategy on soil in its requirement for the protection of soil quality implies that mixed waste composting is undesirable.

In addition, our model does not attribute an external *benefit* to any of the following positive impacts associated with the use of compost:

- Benefits from conservation of biodiversity through avoiding peat use;
- Reduced requirement for liming;
- Reduced susceptibility to soil erosion;
- Improved infiltration (including reduced irrigation requirement and reduced risk of flooding);
- Improved tilth;
- The bioremediation of soil using compost.

Some of these are potentially significant (e.g. the impacts associated with soil erosion occurring beyond the farm gate). However, for many of the benefits outlined above (such as the reduced requirement for liming) it is difficult to estimate the incremental improvements in soil quality that will result from individual compost applications. In other cases (such as the wider benefits associated with a reduction in peat extraction), the environmental costs are difficult to quantify in monetary terms.

On balance, therefore, our model is likely to underestimate the benefits associated with the use of quality compost produced from well-managed facilities. Nonetheless, it can be seen from the results presented in Table 7-12 that the beneficial aspects of compost application remain significant, notwithstanding the omission of those benefits described above.

7.6 Anaerobic Digestion

7.6.1 Financial Costs

Figure 7-10 examines the total financial cost (social metric) per tonne as it applies to an AD plant producing electricity only. Figure 7-11 depicts the private metric. These figures are a graphical representation of the spread between the various elements of cost, including capital and operational expenditure. These are the costs which explain the variation across member states.

The unit capital and operating cost vary in accordance with labour cost as per the methodology in the Annex E. These costs apply to both the social and private metric.

The annualised capex cost is a function of the discount rate, assumed to be 12% for this facility in the private cost metric, over the lifetime of the facility. This is one of the key figures that differs between the private and social cost metrics, the other main factor of course being subsidies, taxes etc. Maintenance has been calculated as a percentage of the unit capital cost.

The revenue generated from energy is comprised of the sale of electricity in this case and the related support mechanism. In order to calculate the social cost metric (at a 4% discount rate) the support mechanism is excluded and revenue generated from energy is calculated on the basis of sales alone (see Figure 7-10).

Electricity sales are priced at the level at which electricity is sold in each member state and this is multiplied by the per tonne generation of electricity for this facility. As previously discussed, renewable heat support offered in each member state is also accounted for in the private cost metric.

The divergence between each member state is explained by the interplay of these key variables. The assumptions that have been made with regard to the capital and operational expenditure per tonne are available in the Annex E. A full list of the data gathered with regard to the support schemes and electricity prices used is available in the Appendix.

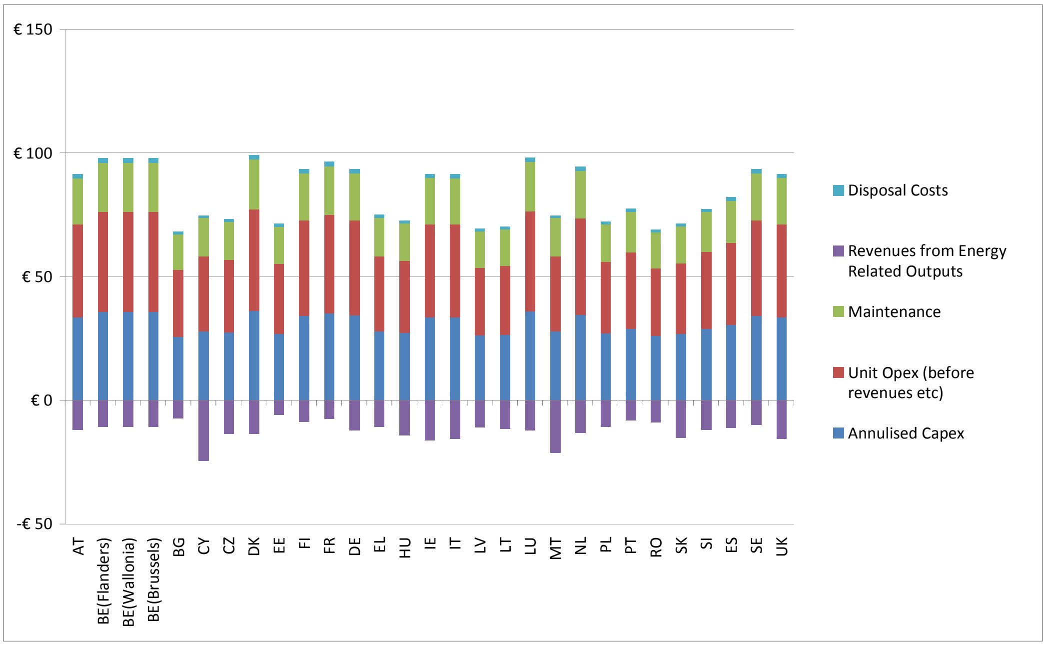
Figure 7-12 examines the variation in cost for an AD facility that is selling biogas as a vehicle fuel. The rules that have been discussed regarding the social and private cost metrics for AD with electricity only apply to the costs for this facility also. Revenue from the sale of biogas as vehicle fuel has been varied per member state and extracted from the European National Gas Vehicle Association.[78]

Figure 7-13 examines the cost variation for an AD facility that is selling biogas to the national grid. The revenue from these activities has been extracted from Eurostat data.[79] Where no revenue is attributed to the sale of gas to the grid in a particular country this is as a result of no grid existing within the member state. The figure examines the social cost and does not include the revenues from support mechanisms which exist is several of the member states. The support mechanisms related to the sale of gas to grid have been detailed in Annex E.

Table 7-13 summarises the costs for each configuration of AD for both of the financial cost metrics. The variation between member states is determined largely by the assumptions made in relation to the labour element of capital and operating costs, but is also significantly influenced by the revenues from sales of energy.

In the private cost metric, these are influenced by support mechanisms for renewable energy generation. This, alongside the higher discount rate, explains the difference between the social and private metrics.

Figure 7-10: Breakdown of Financial Costs for AD: electricity generation only (Social Metric)



[78] NGVA Europe (2009) *Comparison of fuel prices in Europe*, April 2009
[79] Eurostat (2009) Energy Statistics Database. Available at http://epp.eurostat.ec.europa.eu/portal/page/portal/energy/data/database

Figure 7-11: Breakdown of Financial Costs for AD: electricity generation only (Private Metric)

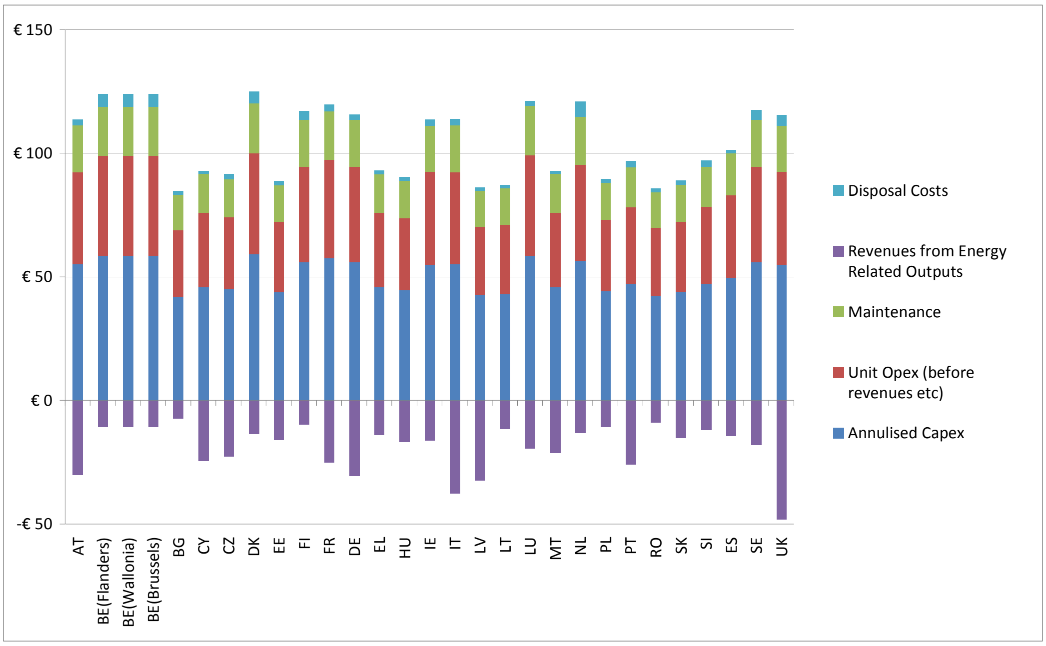


Figure 7-12: Breakdown of Financial Costs for AD: biogas used as a vehicle fuel (Social Metric)

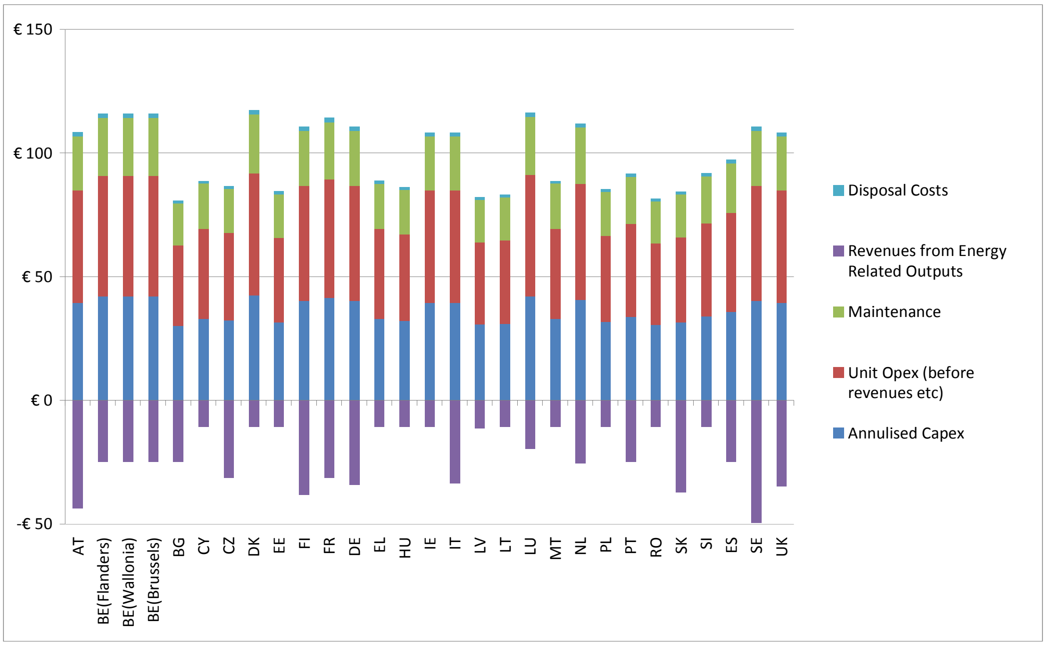


Figure 7-13: Breakdown of Financial Costs for AD: biogas injected into grid (Social Metric)

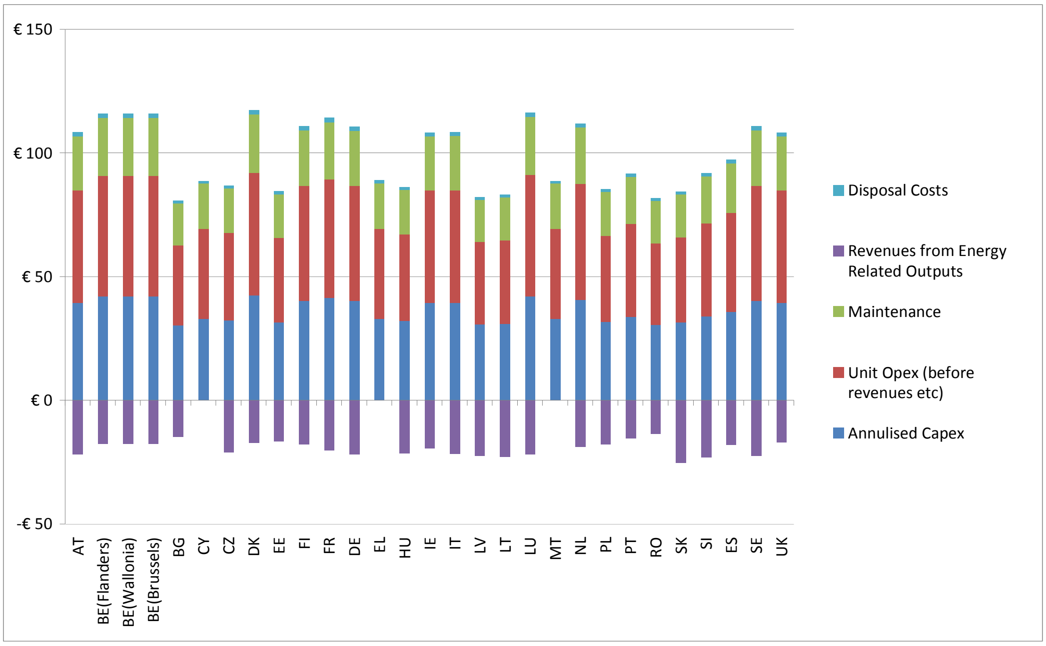


Table 7-13: Financial Costs for AD Variants (Social and Private Cost Metrics)

Country	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI	FR	DE	EL
AD - elec (Social)	€80.0	€87.5	€87.5	€87.5	€61.2	€50.4	€60.0	€85.8	€65.8	€85.2	€89.3	€81.6	€64.6
AD - elec (Private)	€83.7	€113.5	€113.5	€113.5	€77.7	€68.5	€69.2	€111.6	€73.1	€107.6	€94.9	€85.2	€79.4
AD - CHP (Social)	€89.7	€99.5	€99.5	€99.5	€70.6	€63.5	€68.7	€95.7	€75.8	€97.8	€101.0	€92.0	€77.7
AD - CHP (Private)	€99.3	€131.8	€131.8	€131.8	€91.7	€86.4	€82.7	€127.5	€87.7	€126.2	€112.7	€101.6	€97.3
AD - vehicle (Social)	€65.1	€91.4	€91.4	€91.4	€56.2	€78.0	€55.7	€107.0	€74.1	€72.7	€83.3	€76.8	€78.4
AD - vehicle (Private)	€90.8	€121.4	€121.4	€121.4	€75.6	€99.2	€77.2	€136.7	€94.4	€100.0	€110.3	€102.6	€99.6
AD - grid (Social)	€86.9	€98.7	€98.7	€98.7	€66.3	€88.6	€65.9	€100.4	€68.3	€93.1	€94.5	€89.2	€89.0
AD - grid (Private)	€112.7	€128.6	€128.6	€128.6	€85.7	€109.8	€87.5	€130.2	€88.6	€120.4	€121.5	€115.0	€110.2

Country	HU	IE	IT	LV	LT	LU	MT	NL	PL	PT	RO	SK	SI	ES	SE	UK
AD - elec (Social)	€58.9	€75.5	€76.2	€58.6	€58.7	€86.4	€53.8	€81.6	€61.7	€69.5	€60.4	€56.5	€65.8	€71.2	€83.8	€76.1
AD - elec (Private)	€73.8	€97.9	€76.4	€54.2	€75.7	€102.1	€71.9	€108.1	€79.2	€71.4	€77.1	€74.2	€85.4	€87.2	€99.8	€67.4
AD - CHP (Social)	€67.8	€88.5	€85.2	€67.2	€67.3	€97.6	€66.9	€93.1	€71.2	€82.9	€70.1	€65.5	€75.6	€85.4	€94.3	€89.1
AD - CHP (Private)	€87.4	€116.8	€91.2	€67.4	€88.9	€119.5	€89.8	€125.6	€93.4	€89.8	€91.3	€87.8	€100.2	€106.7	€116.3	€86.3
AD - vehicle (Social)	€75.7	€97.8	€75.0	€71.1	€72.6	€97.1	€78.1	€86.8	€75.0	€67.1	€71.2	€47.6	€81.3	€72.6	€61.5	€73.7
AD - vehicle (Private)	€96.2	€123.9	€101.0	€90.8	€92.5	€123.9	€99.3	€117.1	€95.5	€89.8	€90.8	€68.2	€104.0	€95.2	€89.3	€101.4
AD - grid (Social)	€64.8	€89.2	€86.8	€59.9	€60.6	€94.9	€88.7	€93.3	€68.1	€76.5	€68.3	€59.4	€69.0	€79.5	€88.6	€91.5
AD - grid (Private)	€85.4	€115.3	€112.8	€79.6	€80.5	€121.7	€109.9	€123.6	€88.5	€99.2	€87.9	€80.1	€91.7	€102.0	€116.3	€119.2

7.6.2 Environmental Impacts

Our model considers four options for the use of the biogas generated by the digestion process:

1. Combusted on the site of the facility in a gas engine, and used to generate electricity only
2. Combusted on the site of the facility in a gas engine, and used to generate electricity and heat
3. The upgraded biogas is used as a fuel for buses (as is currently the case in Sweden and France), displacing the use of diesel
4. The upgraded biogas is injected into the gas grid

Indicative external damage costs for each of these four options are presented in Table 7-14 using the results for the Czech Republic, as was the case for the in-vessel composting impacts. Further details can be found in Annex F.

The damage costs relate to the following environmental impacts resulting from the anaerobic digestion process:

- Direct climate change emissions from the process are principally biogenic CO_2 emissions. For the on site generation options and the gas to grid option, these relate to the CO_2 released when the biogas is combusted for energy.
- Direct air quality emissions from the process principally relate to NOx emissions as the biogas is burnt. These are lower for the gas to grid option as biogas goes through an additional cleaning process prior to grid injection. The cleaning process removes the NH_3 and H_2S which are a principal cause of NOx and SOx emissions when the biogas is combusted onsite in a gas engine.
- Emissions from burning the cleaned biogas are taken into account within the avoided emissions calculation for the vehicle fuel option. Again, the clean-up of the gas prior to its use within the vehicle reduces the emissions potential.
- Since the plant supplies its own energy, no direct emissions are seen for energy use (these are effectively taken into account through a reduction in the offset energy emissions).
- Offset emissions for the generation of heat at the CHP facility are relatively small as 20% of the heat generated is used in the process, and only 60% of that which is available is assumed to be utilised. In addition, heat generation results in less carbon emissions per unit of energy generated when compared to electricity generation, further reducing the size of the climate change offset.
- The options which generate heat and electrical energy perform favourably as a result of the greater quantity of offset air pollution. The use of diesel in vehicles principally results in some emissions of NOx and relatively small amounts of NMVOC and PM10. However, both and electricity and heat generation in the Czech Republic also result in relatively large emissions of SOx, as the country is heavily reliant on coal and oil within its energy mix.[80] The damage cost associated with SOx emissions dominate the external costs associated with energy generation in the Czech

[80] These impacts are reduced for countries that generate a significant proportion of their electricity from nuclear, gas, hydro-power or wind, and for countries that obtain a significant proportion of their heat from gas

Republic. This partly reflects the relatively low damage cost associated with NOx pollution in the country.

Table 7-14: Indicative External Damage Costs for Anaerobic Digestion

		Climate change	Air quality	Other impacts	Totals
ELECTRICITY ONLY	Process				
	Direct emissions	€10.25	€6.50		€16.75
	Energy use (electricity & diesel)	€0.00	€0.00		€0.00
	Avoided emissions, energy generation				
	Electricity	- €3.24	- €26.19		- €29.43
	Use of compost			-€10.01	- €10.01
	FINAL TOTALS	€5.60	- €18.17	-€10.01	- €22.58
C H P	Process				
	Direct emissions	€10.25	€6.50		€16.75
	Energy use (electricity & diesel)	€0.00	€0.00		€0.00
	Avoided emissions, energy generation				
	Electricity	- €3.24	- €26.19		- €29.43
	Heat	- €0.43	- €5.28		- €5.71
	Use of compost			-€10.01	- €10.01
	FINAL TOTALS	€5.17	- €20.69	-€10.01	- €22.24
AS VEHICLE FUEL	Process				
	Direct emissions	€10.40	€7.06		€17.46
	Energy use (electricity & diesel)	€0.00	€0.00		€0.00
	Avoided emissions, energy generation				
	Diesel	- €3.03	- €13.62		- €16.65
	Use of compost			-€10.01	- €10.01
	FINAL TOTALS	€2.28	- €5.14	-€10.01	- €12.87
GAS INJECTION TO GRID	Process				
	Direct emissions	€10.61	€2.66		€13.27
	Energy use (electricity & diesel)	€0.00	€0.00		€0.00
	Avoided emissions, energy generation				
	Heat	- €3.03	- €32.04		- €35.07
	Use of compost			-€10.01	- €10.01
	FINAL TOTALS	€6.40	- €20.81	-€10.01	- €24.42

Notes:
The energy used at the plant is assumed to be generated by the plant itself (i.e. no grid electricity or heat is required). Avoided emissions associated with energy generation have been adjusted accordingly.
Direct emissions include those associated with the combustion of the biogas to generate energy, and those associated with the burning of the gas in a bus (where it is used as a vehicle fuel).

Section 7.6.2.1 confirms the impacts that are included within the damage costs presented in Table 7-14, whilst Section 7.6.2.2 identifies impacts that have not been included within the monetised damages presented above.

7.6.2.1 Monetised Environmental Impacts

Impacts are calculated on the basis of one tonne of food waste to the process and include CO_2 emissions originating from biogenic sources of carbon.

The anaerobic digestion facility is assumed to produce biogas which can be subsequently used in the following ways:[81]

- Combusted on-site in a gas engine and used to generate only electricity with a gross generation efficiency of 40%;
- Combusted on-site in a gas engine and used to generate electricity and heat, assuming that 60% of the heat can be utilised (gross generation efficiencies are 40% and 45%, respectively);
- Following upgrading of the biogas, it is subsequently used to fuel buses (displacing the use of diesel);
- Following upgrading of the biogas, it is subsequently injected into a gas grid where it is assumed to be used for heat generation.

The anaerobic digestion plant is assumed to supply its own electricity and heat, accounting for 10% and 20% of the initial biogas energy respectively. 3% of the biogas CH_4 content is assumed to be lost through fugitive emissions; a further 2% of the CH_4 is assumed to be lost where the gas is upgraded, with the upgrading process assumed to consume 0.2 kWh per kWh of biogas energy.

The energy generated on-site is assumed to displace equivalent quantities of electricity and heat that would otherwise be generated by other means. The displaced energy sources are assumed to vary from country to country, based on the average fuel mix for electricity and heat generation within that country.[82] Since electricity generated from nuclear power and some renewables (such as hydroelectricity) results in far fewer emissions to air in comparison to that generated from coal or oil, the relative performance of the different types of anaerobic digestion facility will vary between countries depending on the fuel mix of that country.[83]

We assume that the solid residue from the anaerobic digestion plant is subsequently composted and that it undergoes the same process to that previously described for the in-vessel composting process (including the attribution of benefits from the use of compost). This process produces 300 kg of compost per tonne of material entering the AD facility. 90% of the composted residue from anaerobic digestion plant is assumed to be used as a replacement for synthetic fertilisers used within agriculture, with the remaining 10% used to displace peat for horticultural applications and amateur gardening. Although a smaller volume of composted material is produced from the anaerobic digestion process, the displacement of synthetic fertiliser is associated with greater environmental benefits. As such, the overall environmental benefit associated with the use of the composted digestate is greater than that resulting from the use of the compost from IVC and windrow facilities.

7.6.2.2 Non Monetised Environmental Impacts

The non monetised environmental impacts for the anaerobic digestion plant are largely the same as was described for the in-vessel composting process in Section 7.5.2.2.

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81 Generation efficiencies and energy use data are consistent with those reported at the 17th Annual Convention of Fachverband Biogas e.V., January 15-17th 2008, Nuremberg

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

83 Emissions to air from the different energy generation sources are taken from the EcoInvent database v2.1, available from http://www.ecoinvent.ch

7.7 In-Vessel Composting vs. Anaerobic Digestion

The COWI report suggested that switches to composting are more advantageous than switches to anaerobic digestion, when viewed from an economic perspective. The greater environmental benefits of anaerobic digestion were not shown to outweigh the higher treatment costs when compared to composting. However, technologies and market conditions have moved on since this study. Furthermore, environmental damage costs and benefits are being continually updated (though it must be acknowledged that assessment methodologies are running to stand still in maintaining pace with scientific understanding).

The results presented in this section aim to show which biowaste treatment technology provides the lowest cost option for each Member State. The section combines the data on financial cost and environmental damage cost for the biowaste treatment options, and uses this to develop a net cost or benefit for IVC and each of the AD options.

Table 7-15 shows a breakdown of the financial and environmental damage costs for each technology option, which are then aggregated to give the net costs and benefits for each biowaste treatment option. These are presented using the social cost metric, used to maintain consistency with the evaluation of the financial cost of the individual technologies presented in the previous sections. The negative costs indicated in the table represent a benefit to the environment and therefore a welfare gain for society. The table then presents the lowest cost biowaste option under both the private and social metric, for comparison purposes.

The following key messages can be drawn from Table 7-15:

- Whilst the environmental costs associated with IVC are consistently positive (representing a cost to society), the AD options generate net benefits as signified by the negative net external damage costs;
- The variation in external costs is heavily dependant on the energy mix and the level of the damage costs attributed to each tonne of pollutant for each country, as was described in the environmental cost sections for the different technologies. Countries that are reliant on coal or oil for their electricity supply will typically see a greater level of benefit from using the biogas produced by the AD process to generate electricity. These benefits may, however, be reduced where the country concerned has relatively low damage costs per tonne of pollution.
- The environmental cost has little impact on the variation in net cost and benefit for IVC between Member States, as these costs do not vary significantly between the different member states in absolute terms. Furthermore, the only source of variation within the financial costs is the cost of labour, as there are no revenues from energy generation or supporting fiscal mechanisms. Consequently, the net social costs and benefits for IVC only vary between €48.3 and €64.9 across the different Member States;
- There is, however, significant variation in the financial and environmental cost of each of the different AD options, in both relative and absolute terms. The highest and lowest financial costs for all the AD options are €107.0 and €47.6. and the range of environmental costs for all the AD options are from €4.4 to -€24.1 respectively. The net social costs and benefits from the different AD options therefore vary between €102.4 and €41.6;

- Variation in the financial cost of AD appears to be a key factor in determining the lowest cost option from the point of view of society. IVC is typically the lowest cost option whereas the total financial cost of each of the AD options tends to be greater than around €50. There are, however exceptions to this:
    - Where renewable energy support schemes provide a significant revenue stream AD can be the more expensive technology under the social metric (before taking support schemes into account) but can turn out to be the lowest cost option from the private perspective.
    - where both damage and labour costs are low, IVC thus has a significantly lower net social cost than AD. Equally, where damage and labour costs are high, the difference in net social costs are much reduced, in certain cases leading to AD being the preferred option.
- AD options are aided under the social cost metric by of lower discount rate on the cost of capital, notwithstanding the loss in renewable energy support mechanisms. Furthermore, the monetised environmental benefits of AD in this case are larger in proportion to the financial costs, in a number of cases tipping the balance in favour of AD. There are 7 Member States where AD is the lowest cost option under the social metric.
- Under the private metric, in the absence of support mechanisms for renewable energy, then the AD options tend to fare less well against IVC. However, electrical renewable energy support mechanisms lead to AD with electricity generation being favoured for a number of countries including EE, IT, LV, PT and UK where IVC is preferred under the social metric. There are 10 Member States where AD for electricity production is the lowest cost option under the social metric, and one additional country where AD to vehicle fuel remains the preferred option.

Figure 7-14: Number of Countries with Lowest Net Cost to Society (Social Cost Metric) for Each Biowaste Treatment Variant (with Belgium split into the three political institutions)

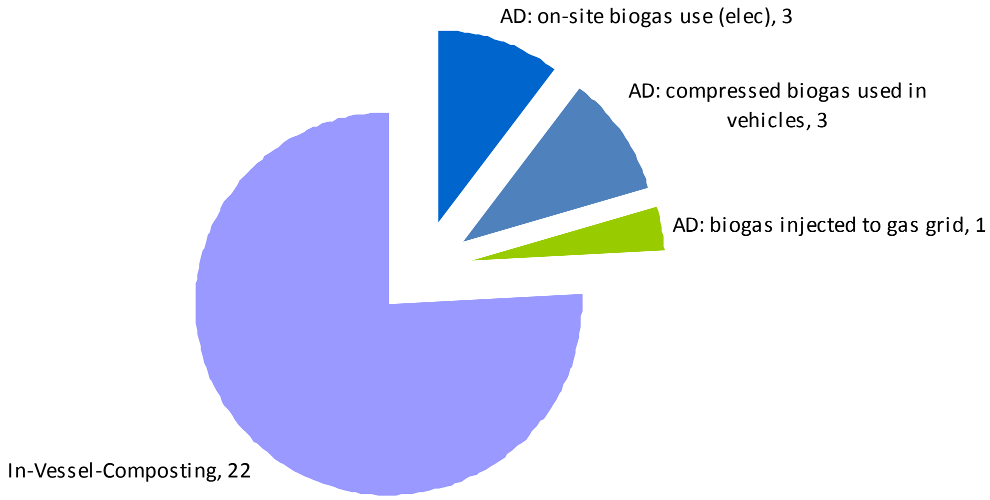


Table 7-15: Comparison of In-vessel Composting and Anaerobic Digestion Approaches in the EU27 Using Social Cost Metric

Technology	Cost	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI
Lowest Cost Option		AD (vehicle)	IVC	IVC	IVC	IVC	AD (elec)	AD (grid)	IVC	IVC	IVC
AD: on-site biogas use (electricity)	Financial	€ 80.0	€ 87.5	€ 87.5	€ 87.5	€ 61.2	€ 50.4	€ 60.0	€ 85.8	€ 65.8	€ 85.2
	External	€ 0.74	-€ 3.04	-€ 3.04	-€ 3.04	-€ 0.72	-€ 7.18	-€ 16.17	-€ 10.19	-€ 6.81	€ 0.14
	Net Social Cost	€ 80.7	€ 84.4	€ 84.4	€ 84.4	€ 60.4	€ 43.2	€ 43.8	€ 75.6	€ 59.0	€ 85.3
AD: on-site biogas use (CHP)	Financial	€ 89.7	€ 99.5	€ 99.5	€ 99.5	€ 70.6	€ 63.5	€ 68.7	€ 95.7	€ 75.8	€ 97.8
	External	-€ 3.08	-€ 7.11	-€ 7.11	-€ 7.11	-€ 1.96	-€ 8.33	-€ 21.05	-€ 12.28	-€ 8.27	-€ 1.03
	Net Social Cost	€ 86.6	€ 92.4	€ 92.4	€ 92.4	€ 68.7	€ 55.1	€ 47.6	€ 83.4	€ 67.5	€ 96.8
AD: biogas used in vehicles	Financial	€ 65.1	€ 91.4	€ 91.4	€ 91.4	€ 56.2	€ 78.0	€ 55.7	€ 107.0	€ 74.1	€ 72.7
	External	-€ 8.90	-€ 6.59	-€ 6.59	-€ 6.59	-€ 0.64	-€ 0.68	-€ 7.74	-€ 4.59	-€ 0.88	-€ 0.80
	Net Social Cost	€ 56.2	€ 84.8	€ 84.8	€ 84.8	€ 55.6	€ 77.3	€ 48.0	€ 102.4	€ 73.2	€ 71.9
AD: biogas injected to gas grid	Financial	€ 86.9	€ 98.7	€ 98.7	€ 98.7	€ 66.3	€ 88.6	€ 65.9	€ 100.4	€ 68.3	€ 93.1
	External	-€ 17.90	-€ 20.40	-€ 20.40	-€ 20.40	-€ 2.70	-€ 2.48	-€ 24.07	-€ 8.60	-€ 4.09	-€ 3.11
	Net Social Cost	€ 69.0	€ 78.3	€ 78.3	€ 78.3	€ 63.6	€ 86.1	€ 41.9	€ 91.8	€ 64.2	€ 90.0
IVC	Financial	€ 57.1	€ 60.5	€ 60.5	€ 60.5	€ 44.3	€ 48.1	€ 47.1	€ 61.2	€ 46.0	€ 58.1
	External	€ 3.35	€ 3.67	€ 3.67	€ 3.67	€ 4.04	€ 3.98	€ 3.56	€ 3.67	€ 3.95	€ 4.03
	Net Social Cost	€ 60.4	€ 64.2	€ 64.2	€ 64.2	€ 48.3	€ 52.1	€ 50.7	€ 64.9	€ 50.0	€ 62.1

Table 7-16: Comparison of In-vessel Composting and Anaerobic Digestion Approaches in the EU27 Using Social Cost Metric (ctd)

Technology	Cost	FR	DE	EL	HU	IE	IT	LV	LT	LU	MT
Lowest Cost Option		IVC	IVC	IVC	IVC	IVC	IVC	IVC	IVC	IVC	AD (elec)
AD: on-site biogas use (electricity)	Financial	€ 89.3	€ 81.6	€ 64.6	€ 58.9	€ 75.5	€ 76.2	€ 58.6	€ 58.7	€ 86.4	€ 53.8
	External	€ 4.31	-€ 15.63	-€ 4.24	-€ 2.52	-€ 7.37	-€ 7.89	€ 2.61	€ 3.07	-€ 0.54	-€ 10.58
	Net Social Cost	€ 93.6	€ 66.0	€ 60.4	€ 56.4	€ 68.2	€ 68.3	€ 61.2	€ 61.8	€ 85.8	€ 43.2
AD: on-site biogas use + CHP	Financial	€ 101.0	€ 92.0	€ 77.7	€ 67.8	€ 88.5	€ 85.2	€ 67.2	€ 67.3	€ 97.6	€ 66.9
	External	€ 0.94	-€ 20.26	-€ 5.53	-€ 4.48	-€ 9.84	-€ 10.48	€ 1.64	€ 1.89	-€ 3.89	-€ 12.00
	Net Social Cost	€ 101.9	€ 71.8	€ 72.1	€ 63.3	€ 78.7	€ 74.7	€ 68.8	€ 69.2	€ 93.7	€ 54.9
AD: compressed biogas used in vehicles	Financial	€ 83.3	€ 76.8	€ 78.4	€ 75.7	€ 97.8	€ 75.0	€ 71.1	€ 72.6	€ 97.1	€ 78.1
	External	-€ 8.10	-€ 10.44	-€ 0.81	-€ 5.49	-€ 4.15	-€ 6.30	-€ 1.42	-€ 1.92	-€ 9.27	-€ 0.90
	Net Social Cost	€ 75.2	€ 66.3	€ 77.6	€ 70.2	€ 93.6	€ 68.7	€ 69.7	€ 70.7	€ 87.9	€ 77.2
AD: biogas injected to gas grid	Financial	€ 94.5	€ 89.2	€ 89.0	€ 64.8	€ 89.2	€ 86.8	€ 59.9	€ 60.6	€ 94.9	€ 88.7
	External	-€ 15.61	-€ 22.98	-€ 3.15	-€ 7.02	-€ 10.16	-€ 10.71	-€ 2.27	-€ 2.85	-€ 14.72	-€ 4.08
	Net Social Cost	€ 78.9	€ 66.2	€ 85.9	€ 57.8	€ 79.0	€ 76.1	€ 57.6	€ 57.8	€ 80.2	€ 84.6
IVC + biofilter	Financial	€ 59.7	€ 58.1	€ 48.1	€ 46.8	€ 57.0	€ 57.1	€ 45.0	€ 45.4	€ 60.7	€ 48.1
	External	€ 3.42	€ 3.13	€ 4.03	€ 3.77	€ 3.58	€ 3.59	€ 4.03	€ 3.95	€ 3.47	€ 4.05
	Net Social Cost	€ 63.1	€ 61.2	€ 52.1	€ 50.6	€ 60.6	€ 60.7	€ 49.0	€ 49.4	€ 64.2	€ 52.1

Table 7-17: Comparison of In-vessel Composting and Anaerobic Digestion Approaches in the EU27 Using Social Cost Metric (ctd)

Technology	Cost	NL	PL	PT	RO	SK	SI	ES	SE	UK
Lowest Cost Option		IVC	AD (elec)	IVC	IVC	AD (vehicle)	IVC	IVC	AD (vehicle)	IVC
AD: on-site biogas use (electricity)	Financial	€ 81.6	€ 61.7	€ 69.5	€ 60.4	€ 56.5	€ 65.8	€ 71.2	€ 83.8	€ 76.1
	External	-€ 14.57	-€ 20.05	-€ 4.88	-€ 1.10	€ 0.37	-€ 5.23	-€ 3.52	€ 4.40	-€ 9.44
	Net Social Cost	€ 67.0	€ 41.6	€ 64.6	€ 59.3	€ 56.9	€ 60.6	€ 67.7	€ 88.2	€ 66.7
AD: on-site biogas use + CHP	Financial	€ 93.1	€ 71.2	€ 82.9	€ 70.1	€ 65.5	€ 75.6	€ 85.4	€ 94.3	€ 89.1
	External	-€ 18.30	-€ 23.85	-€ 6.64	-€ 2.04	-€ 2.29	-€ 8.34	-€ 5.56	€ 2.72	-€ 11.72
	Net Social Cost	€ 74.8	€ 47.3	€ 76.3	€ 68.0	€ 63.2	€ 67.3	€ 79.8	€ 97.0	€ 77.4
AD: compressed biogas used in vehicles	Financial	€ 86.8	€ 75.0	€ 67.1	€ 71.2	€ 47.6	€ 81.3	€ 72.6	€ 61.5	€ 73.7
	External	-€ 8.20	-€ 4.37	-€ 1.70	-€ 0.64	-€ 5.23	-€ 7.00	-€ 3.07	-€ 2.21	-€ 4.47
	Net Social Cost	€ 78.5	€ 70.7	€ 65.4	€ 70.5	€ 42.3	€ 74.3	€ 69.6	€ 59.3	€ 69.3
AD: biogas injected to gas grid	Financial	€ 93.3	€ 68.1	€ 76.5	€ 68.3	€ 59.4	€ 69.0	€ 79.5	€ 88.6	€ 91.5
	External	-€ 18.73	-€ 17.88	-€ 6.83	-€ 1.57	-€ 10.87	-€ 13.73	-€ 7.93	-€ 6.26	-€ 9.74
	Net Social Cost	€ 74.6	€ 50.2	€ 69.7	€ 66.7	€ 48.5	€ 55.2	€ 71.5	€ 82.3	€ 81.7
IVC + biofilter	Financial	€ 58.7	€ 46.5	€ 49.3	€ 44.7	€ 46.1	€ 49.4	€ 51.9	€ 58.2	€ 57.0
	External	€ 3.16	€ 3.66	€ 3.85	€ 4.04	€ 3.87	€ 3.61	€ 3.76	€ 4.02	€ 3.78
	Net Social Cost	€ 61.8	€ 50.2	€ 53.2	€ 48.8	€ 50.0	€ 53.0	€ 55.6	€ 62.2	€ 60.8

Figure 7-15: Net Cost to Society for Different Biowaste Treatment Options (Social Cost Metric), AT to FI

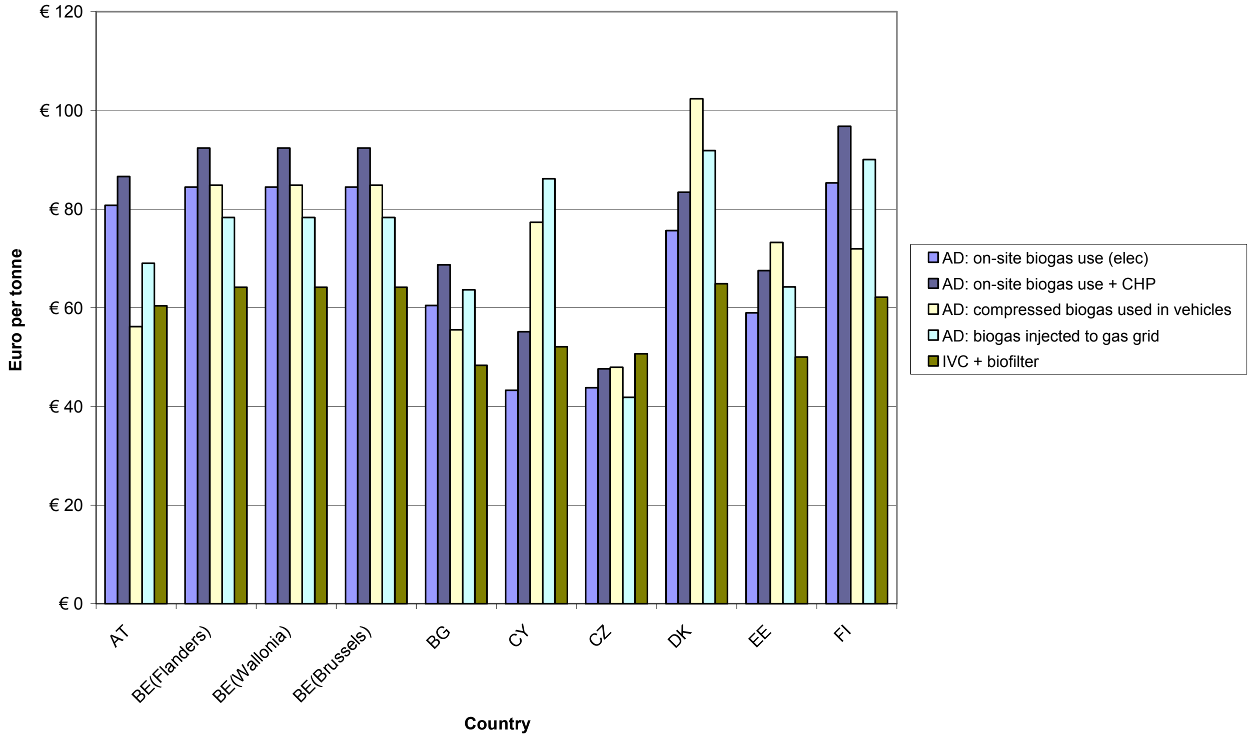


Figure 7-16: Net Cost to Society for Different Biowaste Treatment Options (Social Cost Metric), FR to MT

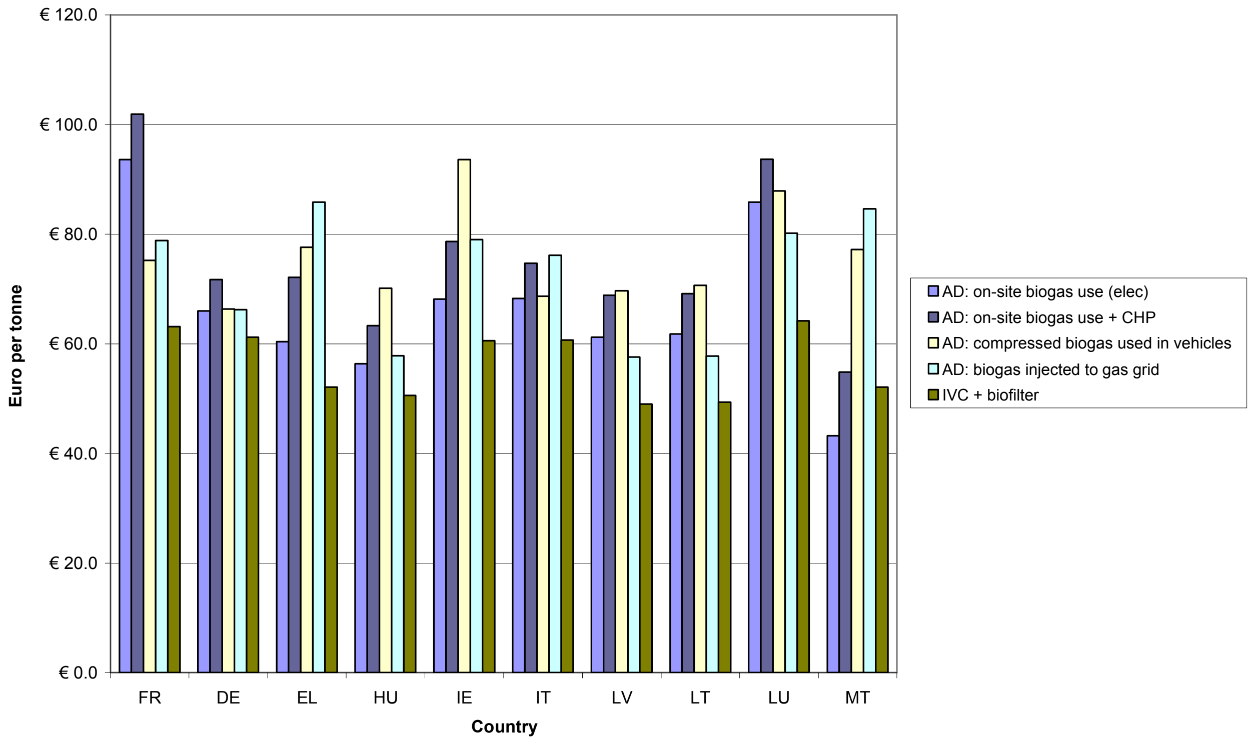
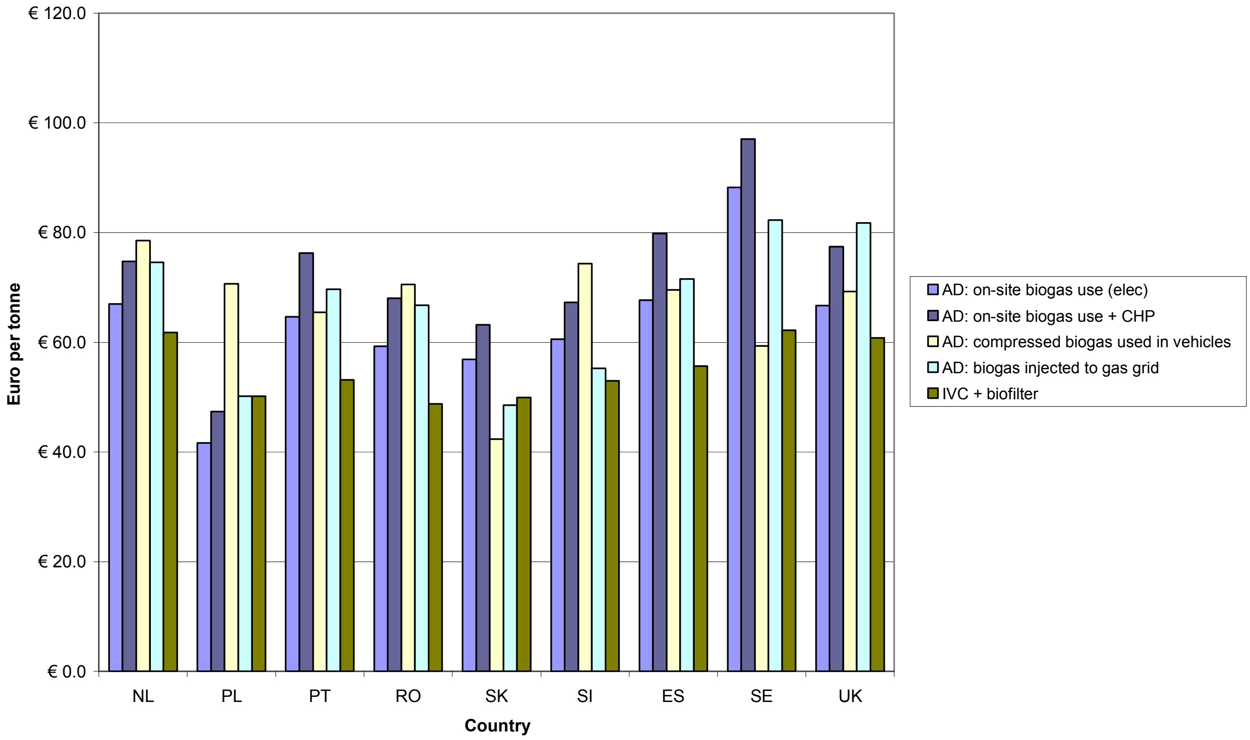


Figure 7-17: Net Cost to Society for Different Biowaste Treatment Options (Social Cost Metric), NL to UK



As Figure 7-15 to Figure 7-17 above show, the differentials between the options vary significantly for some Member States. In Germany, for example, the difference in the net costs to society between all the options is very small, indicating that the actual biowaste treatment process used will not alter the overall benefit cost to society to a significant extent, therefore giving greater flexibility to choose the actual method used to treat biowaste. However, for some Member States the differential is significant, in some cases the highest cost option is twice as costly as the lowest cost option. If an option other than the 'lowest cost' were subsequently chosen (for widespread implementation across a member state), the net benefit to society from switching away from landfill could be greatly reduced. Nevertheless, where these larger ranges are observed, they tend to be a result of a wide difference in costs between the different AD options; the difference between IVC and the lowest of the AD costs in any particular member state is found to be much lower (the most extreme case occurring in Luxemburg where the lowest cost AD option is just 25% more costly than IVC).

What this analysis also shows is that, in the main, the differentials between the lowest and the second lowest cost options, are not significant. This indicates that a Member State could meet the proposed targets set out in the modelling in different ways and not reduce the net benefit to society by a significant factor.

7.8 Incineration

7.8.1 Financial Costs

The financial costs associated with an electricity only incinerator are examined in Table 7-18. This table details the financial costs using the social cost metric and therefore excludes revenues from support schemes and incineration tax as it applies to each country. A discount rate of 4% has been applied to the cost of an incinerator in each member state. The discount rate determines the annualised cost of capital. This has been kept constant across all member states.

Under the social cost metric the countries that have the lowest net costs are those where a) the cost of labour is low (hence capital and operating costs are low) and b) the wholesale price of electricity is high. For incineration electricity only, and under the social metric, France has the highest net cost (€104,3) because is has a low wholesale price of electricity and high labour cost, whereas in Cyprus the low net cost (€57,9) is a result of low labour costs and a significantly higher wholesale electricity price (shown in the chart below).

Under the private cost metric a discount rate of 15% has been used to determine the annualised capital costs. Capital and operating costs will vary across each country as per the indexed labour costs. We have assumed that 16% of the capital costs for an incinerator relate to the labour cost. The labour element of operating costs is assumed to be 21%. The assumptions underlying these figures are available in the Annex E.

The net costs under the private metric are influenced by the same factors as under the social metric, the only difference is the inclusion of support for renewable energy generation and taxes on disposal of residues to landfill. Where countries do have these in place, the net costs can be significantly different to those countries that do not.

In summary the effect of the wholesale electricity price and the variation in labour costs, have the most significant impact on the cost per tonne of incineration across the Member States.

Figure 7-18: Breakdown of Financial Costs for Incineration: Electricity Generation Only (Social Metric)

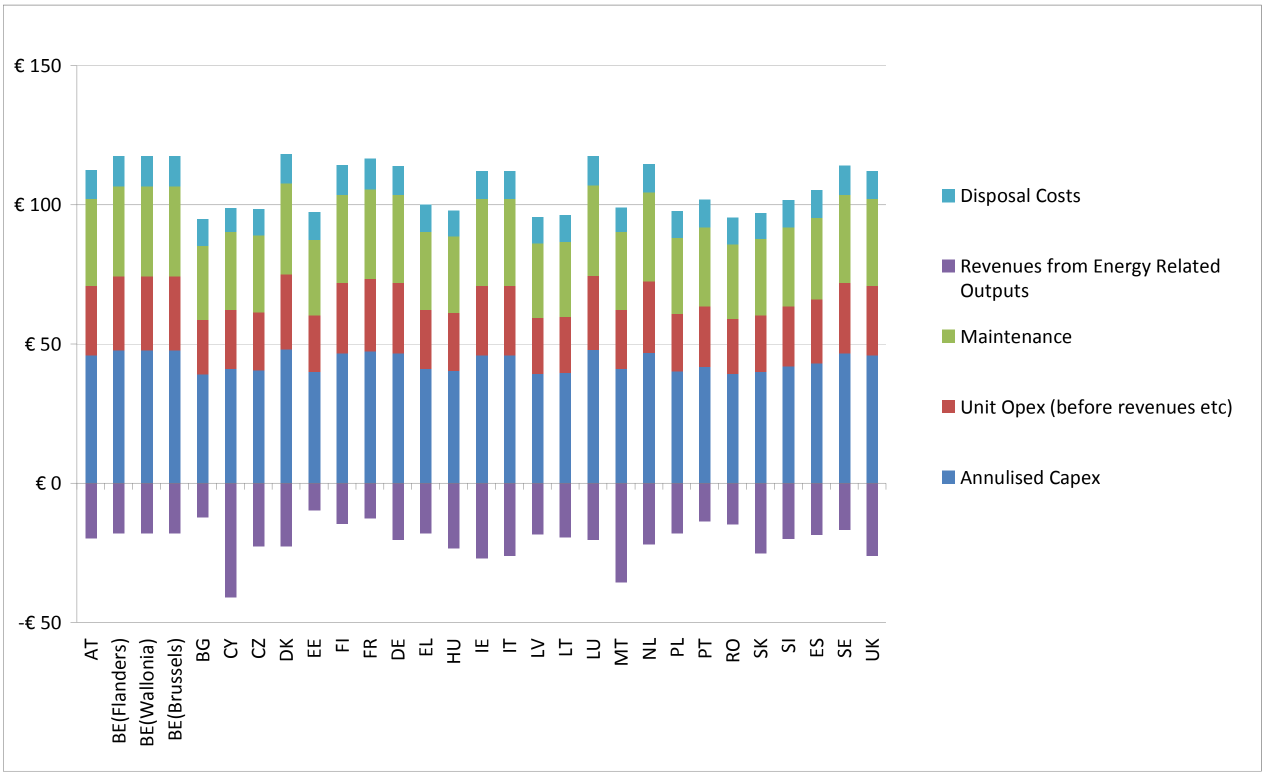


Table 7-18: Net Present Value of Financial Costs for Incineration Variants (Social and Private Cost Metrics)

Country	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI	FR	DE	EL	HU
Incineration (elec only) Social Cost	€ 93.0	€ 99.8	€ 99.8	€ 99.8	€ 83.1	€ 57.9	€ 76.0	€ 95.7	€ 88.0	€ 100.0	€ 104.3	€ 93.9	€ 82.3	€ 74.7
Incineration (elec only) Private Cost	€ 155.0	€ 168.2	€ 164.2	€ 161.2	€ 129.2	€ 106.5	€ 106.2	€ 200.8	€ 118.1	€ 155.5	€ 161.1	€ 148.9	€ 130.9	€ 122.5
Incineration (CHP) Social Cost	€ 93.4	€ 106.1	€ 106.1	€ 106.1	€ 93.3	€ 93.2	€ 84.6	€ 93.8	€ 97.5	€ 110.2	€ 107.4	€ 96.7	€ 106.9	€ 84.8
Incineration (CHP) Private Cost	€ 167.0	€ 186.6	€ 182.6	€ 179.6	€ 149.3	€ 152.1	€ 144.1	€ 209.9	€ 145.5	€ 178.2	€ 169.0	€ 154.0	€ 165.8	€ 142.8
Incineration (heat only) Social Cost	€ 75.8	€ 92.7	€ 92.7	€ 92.7	€ 86.4	€ 111.2	€ 76.9	€ 73.6	€ 89.3	€ 100.2	€ 90.6	€ 81.0	€ 112.5	€ 78.6
Incineration (heat only) Private Cost	€ 140.3	€ 164.0	€ 164.0	€ 164.0	€ 140.6	€ 168.3	€ 134.6	€ 142.5	€ 144.6	€ 167.2	€ 157.2	€ 145.7	€ 169.6	€ 134.8

Country	IE	IT	LV	LT	LU	MT	NL	PL	PT	RO	SK	SI	ES	SE	UK
Incineration (elec only) Social Cost	€ 85.3	€ 86.4	€ 77.4	€ 76.9	€ 97.5	€ 63.7	€ 92.8	€ 80.1	€ 88.6	€ 80.9	€ 71.9	€ 82.0	€ 87.0	€ 97.7	€ 86.4
Incineration (elec only) Private Cost	€ 140.9	€ 140.9	€ 124.0	€ 123.8	€ 153.6	€ 112.3	€ 155.1	€ 127.7	€ 138.0	€ 127.4	€ 115.2	€ 133.0	€ 136.8	€ 191.2	€ 144.4
Incineration (CHP) Social Cost	€ 104.0	€ 86.6	€ 86.1	€ 85.2	€ 100.9	€ 96.5	€ 100.7	€ 90.6	€ 111.4	€ 92.9	€ 83.9	€ 91.7	€ 112.6	€ 99.3	€ 104.6
Incineration (CHP) Private Cost	€ 171.1	€ 152.6	€ 142.6	€ 142.0	€ 169.0	€ 155.4	€ 174.8	€ 148.3	€ 172.8	€ 149.2	€ 141.9	€ 153.2	€ 173.3	€ 210.6	€ 159.4
Incineration (heat only) Social Cost	€ 103.4	€ 69.5	€ 78.3	€ 77.1	€ 85.1	€ 111.5	€ 89.6	€ 84.1	€ 114.7	€ 87.7	€ 79.7	€ 84.0	€ 118.3	€ 82.0	€ 103.5
Incineration (heat only) Private Cost	€ 168.5	€ 133.5	€ 133.1	€ 132.2	€ 151.0	€ 168.6	€ 161.6	€ 140.0	€ 174.2	€ 142.2	€ 135.9	€ 143.7	€ 177.1	€ 149.4	€ 171.0

Table 7-19: Monetised Environmental Impacts for Incineration (food waste)

Country	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI	FR	DE	EL	HU
External costs	€14.5	€5.1	€5.1	€5.1	€9.3	-€0.9	€11.1	€8.8	€0.8	€11.2	€46.0	-€12.1	€4.3	€24.5

Country	IE	IT	LV	LT	LU	MT	NL	PL	PT	RO	SK	SI	ES	SE	UK
External costs	€ 12.0	€ 18.9	€ 17.4	€ 20.0	€ 43.7	-€ 5.5	-€ 12.8	-€ 8.2	€ 6.6	€ 8.6	€ 28.5	€ 25.9	€ 14.1	€ 23.3	€ 10.0

7.8.2 Environmental Impacts

Our model considers two types of incineration facility:

1. A facility generating only electricity;
2. A facility generating electricity and heat (60% of which can be utilised).

Indicative external damage costs for these options are presented in Table 7-14 using the results for the Czech Republic, as was the case for the in-vessel composting impacts.

Table 7-20: Indicative External Damage Costs for Incineration Facilities

		Climate change	Air quality	Other impacts	Totals
ELECTRICITY ONLY	Process				
	Direct emissions	€11.28	€30.02		€41.30
	Energy use (electricity & diesel)	€1.95	€17.58		€19.53
	Avoided emissions, energy generation				
	Electricity	- €5.75	- €43.88		- €49.63
	FINAL TOTALS	€7.48	€3.72		€11.20
C H P	Process				
	Direct emissions	€11.28	€30.02		€41.30
	Energy use (electricity & diesel)	€1.95	€17.58		€19.53
	Avoided emissions, energy generation				
	Electricity	- €3.08	- €23.50		- €26.58
	Heat	- €3.17	- €18.45		- €21.62
	FINAL TOTALS	€6.98	€5.65		€12.63
HEAT ONY	Process				
	Direct emissions	€11.28	€30.02		€41.30
	Energy use (electricity & diesel)	€1.95	€17.58		€19.53
	Avoided emissions, energy generation				
	Heat	- €6.03	- €34.85		- €40.88
	FINAL TOTALS	€7.20	€12.75		€19.95

The damage costs relate to the following environmental impacts resulting from the incineration facility:

- Direct emissions principally relate to the release of biogenic CO_2 as the waste is combusted.
- Air quality impacts resulting from the process are much higher than for the other waste management routes.
- Energy use is significant at incineration facilities, resulting in both climate change and air quality impacts. Grid electricity is used principally within the air pollution control system, and some diesel is used to start the combustion process.
- A considerable proportion of the climate change and air quality impact is however offset by energy generation at the facility. A significant proportion of this results from offset SOx emissions relating to the use of coal and oil within the energy mix of the

country. Some NOx emissions are also offset, although these account for a smaller proportion of the total damage costs.[84]

Section 7.8.2.1 confirms the impacts that are included within the damage costs presented in Table 7-20, whilst Section 7.8.2.2 identifies impacts that have not been included within the monetised damages presented above.

7.8.2.1 Monetised Environmental Impacts

The incineration facility is assumed to generate energy as follows:[85]

- Electricity only with a gross generation efficiency of 28%;
- Electricity and heat (60% of which can be utilised), with gross generation efficiencies of 15% and 40% respectively.

Food and garden waste are modelled with a lower heating value on an as received basis of 5.73 and 6.10 GJ / tonne respectively.

As is the case with the anaerobic digestion plant, incineration facilities are assumed to offset the generation of electricity and heat that would otherwise have been produced by other means. The displaced energy sources are based on the average fuel mix for electricity and heat generation within each country. A forward looking study such as this one probably ought to take into account what is happening at the margin in the relevant energy markets in future years. It would have been preferable to assume that the displacement affects were in relation to sources of electricity and heat that will be displaced at the margin in future years. However, understanding what this marginal source might be in each of the 27 Member States is problematic. Using the average mix is more straight forward, but will almost certainly overstate the benefits in terms of avoided greenhouse gas emissions since most countries are seeking to reduce the carbon intensity of their fuel mix on average, and therefore especially, at the margin.

The incineration plant is assumed to use 78 kWh of electricity and 4.7 litres of diesel per tonne of waste treated at the facility.[86]

For the majority of countries we assume that the incinerator meets the requirements of the waste incineration directive with respect to the reduction of air pollution impacts. In practice, this assumes the incinerator makes use of selective non catalytic reduction technology to reduce NOx emissions which are typically the most significant contributor to the total external costs that are attributed to air pollution.

In Germany, the Netherlands, Belgium and Austria, the use of selective catalytic reduction technology in incineration facilities is commonplace, resulting in improved reduction in the emissions of SOx and NOx from such facilities. We have therefore reduced the direct air quality impacts for incinerators operating in these countries.

The outputs from the incineration process include bottom ash (produced from the un-combustible material) and fly ash which results from the air pollution control residues. We assume that 50% of the former is recycled (to be used as aggregate) and the remainder landfilled. The latter is also landfilled at a hazardous waste landfill site.

[84] These impacts are reduced for countries that generate a significant proportion of their electricity from nuclear, gas, hydro-power or wind, and countries that obtain much of their heat from gas

[85] Efficiencies are towards the upper end of the range of those seen in a survey of European Incinerators updated in 2006. See: I Riemann (2006) CEWEP Energy Report (Status 2001-2004): Results of Specific Data for Energy, Efficiency Rates and Coefficients, Plant Efficiency Factors and NCV of 97 European W-t-E Plants and Determination of the Main Energy Results, updated July 2006

[86] Average energy requirement for facilities using a range of abatement technologies.

The monetised environmental emissions associated with the incineration of food waste are shown in Table 7-19 above.

7.8.2.2 Non Monetised Environmental Impacts

Our analysis does not consider the following impacts:

- The time dependent release of emissions associated with the landfilling of pollution control residues
- A financial estimation of the disamenity associated with living close to an incinerator.

In both cases, only limited data is available upon which to make an estimation of the damage costs.

7.9 Landfill

It must be noted that for modelling purposes we assume all landfills to be compliant with the requirements of the Landfill Directive.

7.9.1 Financial Costs

The financial costs associated with landfill for each of the Member States are shown in Table 7-21. This table details the social cost metric and therefore would exclude any revenue from the support schemes for generation of electricity from landfill gas, and any landfill taxes. The discount rate, for the social metric, is 4%. This is kept constant across all Member States.

Capital and operating costs will vary across each country as per the indexed labour costs. We have assumed that 20% of the capital costs for a landfill relate to the labour cost. The labour element of operating costs is assumed to be 50%. The assumptions underlying these figures are available in the Appendix.

Under the social metric the main determinants of the net present value are the cost of labour and wholesale electricity price. The latter being significant as energy is generated from landfill gas capture. As per the discussion for incineration France has the highest net cost (€38,5) because is has a low wholesale price of electricity and high labour cost, whereas in Cyprus the low net cost (€17,2) is a result of low labour costs and a significantly higher wholesale electricity price.

Table 7-22 below shows the costs under the private metric. The discount rate, representing the weighted average cost of capital for the private metric, is 10%. The key variables are the level of landfill tax, where such taxes exist, across Member States and the level of support for renewable energy generation. Taking the highest and lowest cost figures we find the Netherlands (€125,1) and Cyprus (€22.2). The Netherlands has a very high rate of landfill tax (€85/tonne) while Cyprus has no landfill tax, very high electricity prices that would be received from generation of electricity from landfill gas.

Table 7-21: Financial Costs for Landfill (Social Metric)

Country	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI	FR	DE	EL	HU
Unit Capex	€ 145.1	€ 152.4	€ 152.4	€ 152.4	€ 117.8	€ 125.9	€ 123.9	€ 153.9	€ 121.5	€ 147.3	€ 150.7	€ 147.3	€ 125.9	€ 123.3
Discount Rate	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%
Annualised Capex	€ 15.5	€ 16.2	€ 16.2	€ 16.2	€ 12.6	€ 13.4	€ 13.2	€ 16.4	€ 12.9	€ 15.7	€ 16.1	€ 15.7	€ 13.4	€ 13.1
Unit Opex	€ 8.8	€ 9.9	€ 9.9	€ 9.9	€ 4.7	€ 5.9	€ 5.6	€ 10.1	€ 5.2	€ 9.1	€ 9.6	€ 9.1	€ 5.9	€ 5.5
Maintenance	€ 7.3	€ 7.6	€ 7.6	€ 7.6	€ 5.9	€ 6.3	€ 6.2	€ 7.7	€ 6.1	€ 7.4	€ 7.5	€ 7.4	€ 6.3	€ 6.2
Energy Revenues	€ 6.9	€ 6.2	€ 6.2	€ 6.2	€ 4.2	€ 14.3	€ 7.8	€ 7.9	€ 3.4	€ 5.0	€ 4.3	€ 7.0	€ 6.2	€ 8.1
Total (NPV / Tonne)	€ 33.4	€ 37.4	€ 37.4	€ 37.4	€ 23.6	€ 17.2	€ 22.7	€ 36.4	€ 26.1	€ 36.2	€ 38.5	€ 34.2	€ 25.3	€ 22.1

Country	IE	IT	LV	LT	LU	MT	NL	PL	PT	RO	SK	SI	ES	SE	UK
Unit Capex	€ 145.0	€ 145.1	€ 119.3	€ 120.2	€ 152.9	€ 125.9	€ 148.5	€ 122.6	€ 128.6	€ 118.8	€ 121.7	€ 128.8	€ 134.0	€ 147.3	€ 145.0
Discount Rate	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%	4%
Annualised Capex	€ 15.5	€ 15.5	€ 12.7	€ 12.8	€ 16.3	€ 13.4	€ 15.8	€ 13.1	€ 13.7	€ 12.7	€ 13.0	€ 13.7	€ 14.3	€ 15.7	€ 15.5
Unit Opex	€ 8.8	€ 8.8	€ 4.9	€ 5.0	€ 9.9	€ 5.9	€ 9.3	€ 5.4	€ 6.3	€ 4.8	€ 5.2	€ 6.3	€ 7.1	€ 9.1	€ 8.8
Maintenance	€ 7.3	€ 7.3	€ 6.0	€ 6.0	€ 7.6	€ 6.3	€ 7.4	€ 6.1	€ 6.4	€ 5.9	€ 6.1	€ 6.4	€ 6.7	€ 7.4	€ 7.3
Revenues from Energy	€ 9.4	€ 9.0	€ 6.4	€ 6.8	€ 7.0	€ 12.3	€ 7.6	€ 6.2	€ 4.7	€ 5.1	€ 8.7	€ 6.9	€ 6.4	€ 5.8	€ 9.0
Total (NPV per Tonne)	€ 30.8	€ 31.2	€ 22.0	€ 22.1	€ 36.8	€ 19.1	€ 34.2	€ 23.7	€ 28.0	€ 23.1	€ 20.8	€ 25.8	€ 28.7	€ 35.5	€ 31.2

Table 7-22: Financial Costs for Landfill (Private Metric)

Country	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI	FR	DE	EL	HU
Unit Capex	€ 145.1	€ 152.4	€ 152.4	€ 152.4	€ 117.8	€ 125.9	€ 123.9	€ 153.9	€ 121.5	€ 147.3	€ 150.7	€ 147.3	€ 125.9	€ 123.3
Discount Rate	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%
Annualised Capex	€ 21.3	€ 22.4	€ 22.4	€ 22.4	€ 17.3	€ 18.5	€ 18.2	€ 22.6	€ 17.8	€ 21.6	€ 22.1	€ 21.6	€ 18.5	€ 18.1
Unit Opex	€ 8.8	€ 9.9	€ 9.9	€ 9.9	€ 4.7	€ 5.9	€ 5.6	€ 10.1	€ 5.2	€ 9.1	€ 9.6	€ 9.1	€ 5.9	€ 5.5
Maintenance	€ 7.3	€ 7.6	€ 7.6	€ 7.6	€ 5.9	€ 6.3	€ 6.2	€ 7.7	€ 6.1	€ 7.4	€ 7.5	€ 7.4	€ 6.3	€ 6.2
Energy Revenues	€ 6.9	€ 6.2	€ 6.2	€ 6.2	€ 4.2	€ 14.3	€ 7.8	€ 7.9	€ 9.3	€ 5.7	€ 4.3	€ 7.0	€ 6.2	€ 8.1
Landfill Tax	€ 8.0	€ 60.0	€ 60.0	€ 60.0	€ 0.0	€ 0.0	€ 16.0	€ 50.0	€ 8.0	€ 30.0	€ 9.2	€ 0.0	€ 0.0	€ 0.0
Total (NPV / Tonne)	€ 47.2	€ 103.5	€ 103.5	€ 103.5	€ 28.3	€ 22.2	€ 43.6	€ 92.6	€ 33.1	€ 71.5	€ 53.7	€ 40.1	€ 30.3	€ 27.1

Country	IE	IT	LV	LT	LU	MT	NL	PL	PT	RO	SK	SI	ES	SE	UK
Unit Capex	€ 145.0	€ 145.1	€ 119.3	€ 120.2	€ 152.9	€ 125.9	€ 148.5	€ 122.6	€ 128.6	€ 118.8	€ 121.7	€ 128.8	€ 134.0	€ 147.3	€ 145.0
Discount Rate	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%	10%
Annualised Capex	€ 21.3	€ 21.3	€ 17.5	€ 17.6	€ 22.4	€ 18.5	€ 21.8	€ 18.0	€ 18.9	€ 17.4	€ 17.9	€ 18.9	€ 19.7	€ 21.6	€ 21.3
Unit Opex	€ 8.8	€ 8.8	€ 4.9	€ 5.0	€ 9.9	€ 5.9	€ 9.3	€ 5.4	€ 6.3	€ 4.8	€ 5.2	€ 6.3	€ 7.1	€ 9.1	€ 8.8
Maintenance	€ 7.3	€ 7.3	€ 6.0	€ 6.0	€ 7.6	€ 6.3	€ 7.4	€ 6.1	€ 6.4	€ 5.9	€ 6.1	€ 6.4	€ 6.7	€ 7.4	€ 7.3
Revenues from Energy	€ 9.4	€ 22.0	€ 6.4	€ 6.8	€ 11.3	€ 12.3	€ 7.6	€ 6.2	€ 4.7	€ 5.1	€ 8.7	€ 6.9	€ 14.0	€ 10.4	€ 11.4
Landfill Tax	€ 15.0	€ 26.0	€ 0.0	€ 0.0	€ 0.0	€ 0.0	€ 85.0	€ 0.0	€ 19.0	€ 0.0	€ 7.9	€ 19.0	€ 0.0	€ 43.0	€ 50.0
Total (NPV per Tonne)	€ 51.7	€ 50.1	€ 26.8	€ 26.9	€ 38.7	€ 24.2	€ 125.1	€ 28.6	€ 52.1	€ 27.8	€ 33.6	€ 50.0	€ 26.5	€ 79.7	€ 84.6

Table 7-23: Monetised Environmental Impacts for Landfill (food waste)

Country	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI	FR	DE	EL	HU
External costs	€ 66.94	€ 76.15	€ 76.15	€ 76.15	€ 50.50	€ 49.56	€ 69.76	€ 58.59	€ 50.35	€ 51.18	€ 66.00	€ 71.63	€ 50.97	€ 62.20

Country	IE	IT	LV	LT	LU	MT	NL	PL	PT	RO	SK	SI	ES	SE	UK
External costs	€ 54.56	€ 62.73	€ 52.78	€ 52.65	€ 75.63	€ 53.98	€ 71.85	€ 58.95	€ 52.73	€ 50.38	€ 64.26	€ 64.96	€ 54.88	€ 56.05	€ 64.57

7.9.2 Environmental Impacts

Indicative external damage costs for the landfilling of biowaste are presented in Table 7-24 using the results for the Czech Republic.

Table 7-24: Indicative External Damage Costs for the Landfill of Food Waste

	Climate change	Air quality	Other impacts	Totals
Process				
Direct emissions (non energy)	€48.45	€18.06		€66.51
Energy use (electricity & diesel)	€0.15	€3.63		€3.78
Avoided emissions, energy generation				
Electricity	- €0.90	- €7.30		- €8.20
FINAL TOTALS	€47.70	€14.39		€62.09

The damage costs relate to the following environmental impacts resulting from the landfilling of food waste:

- Impacts are dominated by the climate change impacts caused by the direct emissions from the process – principally fugitive CH_4 emissions resulting from the anaerobic degradation of carbon in the landfill. Some biogenic CO_2 emissions also occur.
- Air quality impacts resulting from the process are dominated by NH_3 and NOx emissions resulting from the relatively high nitrogen content of food waste.
- The process also generates energy and this offsets some of the climate change and air quality impacts. The offset associated with this generation is reduced as it occurs over the 150 year lifetime of the material within the landfill, and benefits occurring during subsequent years are therefore discounted.

7.9.2.1 Monetised Environmental Impacts

Emissions to air from landfill are assumed to occur over time as the waste degrades, and are modelled using a first order decay model. We attribute different rates of decay to the different carbon fractions occurring in food waste. Whilst the lignin contained in plant-based materials decays relatively slowly, fats and sugars degrade much more rapidly. The monetisation of the time dependent emissions to air considers that impacts occurring in subsequent years are subject to a discount rate of 4%.

The issue of landfill gas capture efficiencies is one which critically influences the performance of landfills in any external cost assessment. We assume that landfills in all member states capture 50% of the gas that results from the biological degradation of the waste, during the 150 year period of the analysis.[87]

Of the gas that is captured, 60% of the gas is used to generate electricity, displacing an equivalent amount of generation which would otherwise have occurred using the average

[87] A review of literature on this subject is provided in Eunomia (2008) Development of Marginal Abatement Cost Curves for the Waste Sector, Report for Committee on Climate Change, Defra and the Environment Agency, December 2008. This is likely to be high for some countries at present. However, we have been asked to assume implementation of the Landfill Directive, in which case, sites should be capturing gas as far as possible and using it to generate energy as far as possible.,

fuel mix of that country. The remaining captured gas is assumed to be flared, which oxidises the CH_4 contained within the gas to CO_2. We further assume that 10% of the CH_4 in the uncaptured gas is oxidised by the covering material of the landfill.

Landfill gas can contain a considerable array of trace elements in addition to its principal constituents CH_4 and CO_2. The time-dependent emission of trace gases, both to the atmosphere and via the captured gas is estimated on a proportional basis, based on the CH_4 content of the gas.

The monetised environmental emissions associated with the landfilling of food waste are shown in Table 7-23 above.

7.9.2.2 Non Monetised Environmental Impacts

Our model does not include external costs associated with the following impacts:

- Emissions of leachate to soil and water;
- Impacts associated with odour;
- A financial estimation of the disamenity associated with living close to a landfill.

These impacts are likely to be relatively small in comparison to that which is attributed to the emissions of greenhouse gas to air.

As was previously indicated in Section 7.9.2.1, landfill gas can include a considerable variety of trace elements. Our estimation of the damage costs associated with the air pollution impacts is however based on a relatively small group of pollutants for which the health impacts are reasonably well understood. It is therefore likely that the model underestimates the impacts associated with emissions to air from landfill.

7.10 Switching from Landfill to Organic Treatment Systems

7.10.1 Cost of Switch

The costs associated with switches are as described in previous sections relating to landfill and IVC/AD technologies. The private costs in **Table 7-25** use discount rates representing the technology specific weighted average cost of capital, and include taxes and subsidies, such as renewable energy support schemes. The social costs **Table 7-28** ignore taxes and subsidies, and use the standard 4% discount rate. The costs for each treatment type show the costs of treating a tonne of biowaste. To calculate the net costs or benefits associated with switching from one treatment route to another, we look at the differences between the treatment costs as outlined in 7.10.3

7.10.2 Environmental Impact of Switch

The external damage costs associated with landfilling food waste are high across all Member States, ranging from €49.46 in Cyprus to €76.15 in Flanders. As a result, the switch from landfill to any of the organic treatment methods is always beneficial from the perspective of the environment, with the major part of the benefit being derived from the removal of the material from landfill.

7.10.3 Net Benefit of Switch

The tables below show the social welfare change when a tonne of waste is switched from landfill, to the different biowaste treatments. We assume that the issue of waste growth associated with free garden waste collections, as discussed in Section 7.4.1, are mitigated by maintaining a marginal incentive to not generate additional biowaste (i.e. through maintaining a marginal charge).

Table 7-25 shows this for all countries for the ’private cost’ metric, **Table 7-28** for the ‘social cost’ metric. The ‘Net Cost of Switch’ figure represents the outcome of the following equation

(RESIDUAL WASTE FINANCIAL COST + RESIDUAL WASTE EXTERNAL COST) *MINUS*

(BIOWASTE FINANCIAL COST + BIOWASTE EXTERNAL COST)

A negative figure implies a reduction in the net financial plus external costs, and therefore a welfare gain.

The best performing switches from landfill for each Member State, and the subsequent ordering of the other switching options, are the same as those outlined in **Table 7-15** where the lowest cost option of the AD variations and IVC are detailed, as the impacts from landfill (for each individual member state) are held constant when compared against each of the organic treatment systems.

A number of factors have a large influence on the extent of the net benefit for individual Member States of moving from landfill to the organic treatment options.

- The costs associated with treatment via landfill are an important factor, so where landfill taxes are high (such as The Netherlands) the net benefit of switches to organic treatment systems will tend to be higher (under the private metric). Likewise, where the costs associated with landfill are low, for example in Cyprus, and where the revenues from electricity generation are particularly high, switches to organic options, especially where no electricity is generated, provide much lower benefits. In the example of AD gas-to-grid for Cyprus, there is effectively no gas grid on Cyprus, so the revenue is stated as zero. Therefore there is a net cost shown of switching from landfill to this organics treatment system. This would obviously not be considered as a realistic option for Cyprus. The net cost associated with the switch from landfill to the gas-to-grid and biogas to vehicle fuel options for Latvia and Estonia similarly result from the very high cost of the gas upgrading options in comparison to the very low cost of landfill in that country.
- The costs of landfill are particularly important when considering the switch from landfill to IVC, as there is little variation in IVC costs between member states. The only source of variation within the financial costs for IVC is the cost of labour, as there are no revenues from energy generation or supporting fiscal mechanisms
- Where electricity generation has a lower carbon intensity, such as in Sweden, the benefits of producing electricity from AD are reduced relative to the production of biogas for vehicle fuel, by comparison with other countries where electricity generation with a higher carbon intensity. Looking ahead, with various policy drivers working to reduce the carbon intensity of electricity generation, it may well be that the use of biogas to replace vehicle fuel becomes increasingly attractive.
- The variation in external benefits from switching to particular organic treatment systems is dependant on the energy mix and the level of the damage costs attributed to each tonne of pollutant for each country, as was described in the

environmental cost sections for the different technologies. Countries that are reliant on coal or oil for their electricity supply (such as the Czech Republic) will typically see a greater level of benefit from using the biogas produced by the AD process to generate electricity. These benefits may, however, be reduced where the country concerned has relatively low damage costs per tonne of pollution.

It is important to note that the vast majority of the switches from landfill provide a net social benefit, highlighting the importance of diverting food waste from landfill.

Figure 7-19: Net Cost to Society from Switch of Biowaste from Landfill to Lowest Cost Option (Social Cost Metric)

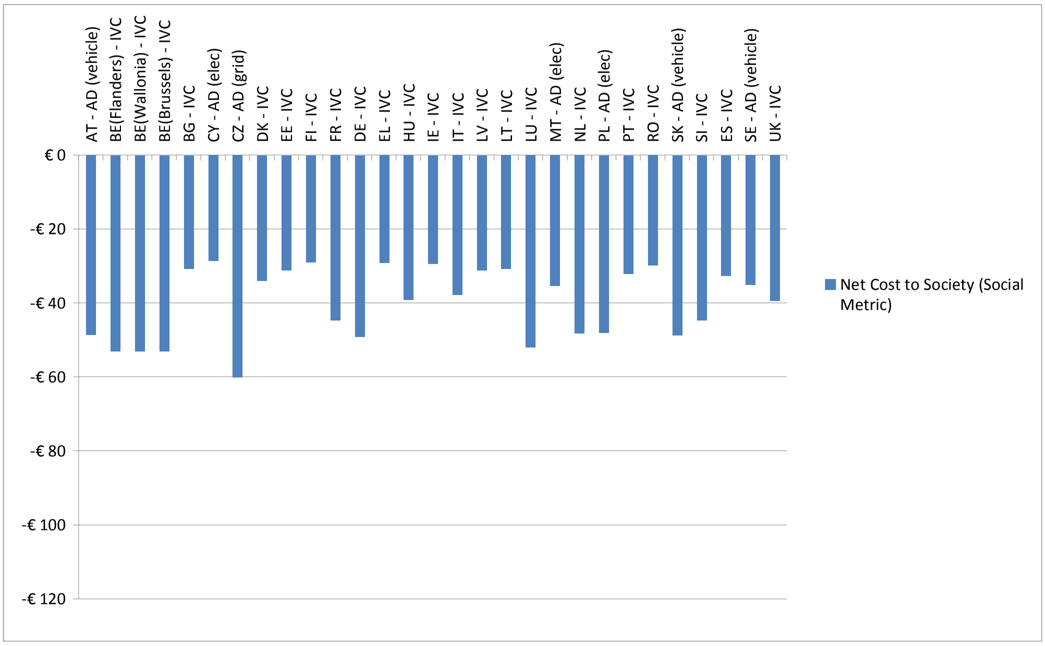


Figure 7-20: Net Cost to Society from Switch of Biowaste from Landfill to Lowest Cost Option (Private Cost Metric)

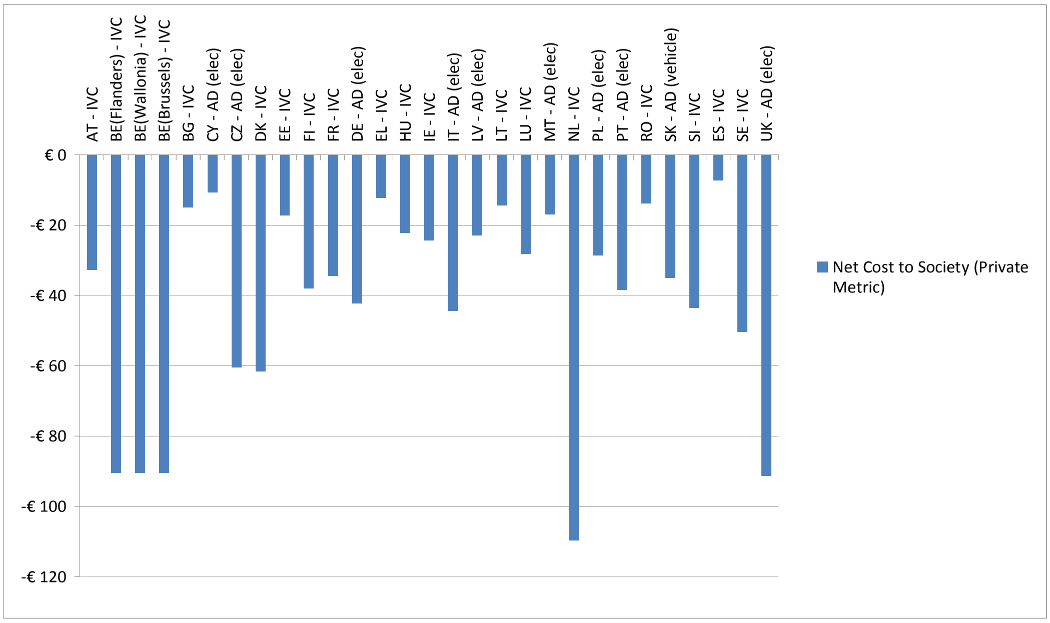


Table 7-25: Switch from Landfill to Different Biowaste Treatments in the EU27 Using Social Metric (external costs included)

Technology	Cost	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI
Landfill to **AD: on-site biogas use (elec)**	From	€ 100.3	€ 113.5	€ 113.5	€ 113.5	€ 74.1	€ 66.7	€ 92.4	€ 95.0	€ 76.4	€ 87.4
	To	€ 80.7	€ 84.4	€ 84.4	€ 84.4	€ 60.4	€ 43.2	€ 43.8	€ 75.6	€ 59.0	€ 85.3
	Net Cost of Switch	-€ 19.58	-€ 29.11	-€ 29.11	-€ 29.11	-€ 13.61	-€ 23.50	-€ 48.66	-€ 19.36	-€ 17.44	-€ 2.12
Landfill to **AD: on-site biogas use (CHP)**	From	€ 100.32	€ 113.54	€ 113.54	€ 113.54	€ 74.05	€ 66.73	€ 92.43	€ 94.99	€ 76.42	€ 87.42
	To	€ 86.6	€ 92.4	€ 92.4	€ 92.4	€ 68.7	€ 55.1	€ 47.6	€ 83.4	€ 67.5	€ 96.8
	Net Cost of Switch	-€ 13.7	-€ 21.1	-€ 21.1	-€ 21.1	-€ 5.4	-€ 11.6	-€ 44.8	-€ 11.6	-€ 8.9	€ 9.4
Landfill to **AD: compressed biogas used in vehicles**	From	€ 100.32	€ 113.54	€ 113.54	€ 113.54	€ 74.05	€ 66.73	€ 92.43	€ 94.99	€ 76.42	€ 87.42
	To	€ 56.2	€ 84.8	€ 84.8	€ 84.8	€ 55.6	€ 77.3	€ 48.0	€ 102.4	€ 73.2	€ 71.9
	Net Cost of Switch	-€ 44.1	-€ 28.7	-€ 28.7	-€ 28.7	-€ 18.5	€ 10.6	-€ 44.5	€ 7.4	-€ 3.2	-€ 15.5
Landfill to **AD: biogas injected to gas grid**	From	€ 100.32	€ 113.54	€ 113.54	€ 113.54	€ 74.05	€ 66.73	€ 92.43	€ 94.99	€ 76.42	€ 87.42
	To	€ 69.0	€ 78.3	€ 78.3	€ 78.3	€ 63.6	€ 86.1	€ 41.9	€ 91.8	€ 64.2	€ 90.0
	Net Cost of Switch	-€ 31.3	-€ 35.3	-€ 35.3	-€ 35.3	-€ 10.4	€ 19.4	-€ 50.6	-€ 3.2	-€ 12.2	€ 2.6
Landfill to **IVC**	From	€ 100.32	€ 113.54	€ 113.54	€ 113.54	€ 74.05	€ 66.73	€ 92.43	€ 94.99	€ 76.42	€ 87.42
	To	€ 60.4	€ 64.2	€ 64.2	€ 64.2	€ 48.3	€ 52.1	€ 50.7	€ 64.9	€ 50.0	€ 62.1
	Net Cost of Switch	-€ 39.9	-€ 49.4	-€ 49.4	-€ 49.4	-€ 25.7	-€ 14.7	-€ 41.8	-€ 30.1	-€ 26.4	-€ 25.3

Table 7-26: Switch from Landfill to Different Biowaste Treatments in the EU27 Using Social Metric (ctd)

Technology	Cost	FR	DE	EL	HU	IE	IT	LV	LT	LU	MT
Landfill to AD: on-site biogas use (elec)	From	€ 104.5	€ 105.9	€ 76.2	€ 84.3	€ 85.4	€ 93.9	€ 74.8	€ 74.7	€ 112.4	€ 73.1
	To	€ 93.6	€ 66.0	€ 60.4	€ 56.4	€ 68.2	€ 68.3	€ 61.2	€ 61.8	€ 85.8	€ 43.2
	Net Cost of Switch	-€ 10.88	-€ 39.87	-€ 15.86	-€ 27.90	-€ 17.24	-€ 25.64	-€ 13.63	-€ 12.95	-€ 26.54	-€ 29.86
Landfill to AD: on-site biogas use + CHP	From	€ 104.48	€ 105.86	€ 76.22	€ 84.30	€ 85.40	€ 93.94	€ 74.81	€ 74.74	€ 112.39	€ 73.09
	To	€ 101.9	€ 71.8	€ 72.1	€ 63.3	€ 78.7	€ 74.7	€ 68.8	€ 69.2	€ 93.7	€ 54.9
	Net Cost of Switch	-€ 2.6	-€ 34.1	-€ 4.1	-€ 21.0	-€ 6.7	-€ 19.3	-€ 6.0	-€ 5.6	-€ 18.7	-€ 18.2
Landfill to AD: compressed biogas used in vehicles	From	€ 104.48	€ 105.86	€ 76.22	€ 84.30	€ 85.40	€ 93.94	€ 74.81	€ 74.74	€ 112.39	€ 73.09
	To	€ 75.2	€ 66.3	€ 77.6	€ 70.2	€ 93.6	€ 68.7	€ 69.7	€ 70.7	€ 87.9	€ 77.2
	Net Cost of Switch	-€ 29.3	-€ 39.5	€ 1.4	-€ 14.1	€ 8.2	-€ 25.3	-€ 5.2	-€ 4.1	-€ 24.5	€ 4.1
Landfill to AD: biogas injected to gas grid	From	€ 104.48	€ 105.86	€ 76.22	€ 84.30	€ 85.40	€ 93.94	€ 74.81	€ 74.74	€ 112.39	€ 73.09
	To	€ 78.9	€ 66.2	€ 85.9	€ 57.8	€ 79.0	€ 76.1	€ 57.6	€ 57.8	€ 80.2	€ 84.6
	Net Cost of Switch	-€ 25.6	-€ 39.6	€ 9.6	-€ 26.5	-€ 6.4	-€ 17.8	-€ 17.2	-€ 17.0	-€ 32.2	€ 11.5
Landfill to IVC	From	€ 104.48	€ 105.86	€ 76.22	€ 84.30	€ 85.40	€ 93.94	€ 74.81	€ 74.74	€ 112.39	€ 73.09
	To	€ 63.1	€ 61.2	€ 52.1	€ 50.6	€ 60.6	€ 60.7	€ 49.0	€ 49.4	€ 64.2	€ 52.1
	Net Cost of Switch	-€ 41.4	-€ 44.6	-€ 24.1	-€ 33.7	-€ 24.8	-€ 33.3	-€ 25.8	-€ 25.4	-€ 48.2	-€ 21.0

Table 7-27: Switch from Landfill to Different Biowaste Treatments in the EU27 Using Social Metric (ctd)

Technology	Cost	NL	PL	PT	RO	SK	SI	ES	SE	UK
Landfill to AD: on-site biogas use (elec)	From	€ 106.0	€ 82.7	€ 80.7	€ 73.4	€ 85.0	€ 90.8	€ 83.6	€ 91.5	€ 95.8
	To	€ 67.0	€ 41.6	€ 64.6	€ 59.3	€ 56.9	€ 60.6	€ 67.7	€ 88.2	€ 66.7
	Net Cost of Switch	-€ 39.02	-€ 41.01	-€ 16.05	-€ 14.19	-€ 28.17	-€ 30.22	-€ 15.93	-€ 3.31	-€ 29.06
Landfill to AD: on-site biogas use + CHP	From	€ 106.00	€ 82.65	€ 80.69	€ 73.44	€ 85.05	€ 90.81	€ 83.62	€ 91.53	€ 95.76
	To	€ 74.8	€ 47.3	€ 76.3	€ 68.0	€ 63.2	€ 67.3	€ 79.8	€ 97.0	€ 77.4
	Net Cost of Switch	-€ 31.3	-€ 35.3	-€ 4.4	-€ 5.4	-€ 21.9	-€ 23.5	-€ 3.8	€ 5.5	-€ 18.3
Landfill to AD: compressed biogas used in vehicles	From	€ 106.00	€ 82.65	€ 80.69	€ 73.44	€ 85.05	€ 90.81	€ 83.62	€ 91.53	€ 95.76
	To	€ 78.5	€ 70.7	€ 65.4	€ 70.5	€ 42.3	€ 74.3	€ 69.6	€ 59.3	€ 69.3
	Net Cost of Switch	-€ 27.5	-€ 12.0	-€ 15.3	-€ 2.9	-€ 42.7	-€ 16.5	-€ 14.1	-€ 32.2	-€ 26.5
Landfill to AD: biogas injected to gas grid	From	€ 106.00	€ 82.65	€ 80.69	€ 73.44	€ 85.05	€ 90.81	€ 83.62	€ 91.53	€ 95.76
	To	€ 74.6	€ 50.2	€ 69.7	€ 66.7	€ 48.5	€ 55.2	€ 71.5	€ 82.3	€ 81.7
	Net Cost of Switch	-€ 31.5	-€ 32.5	-€ 11.0	-€ 6.7	-€ 36.5	-€ 35.6	-€ 12.1	-€ 9.2	-€ 14.0
Landfill to IVC	From	€ 106.00	€ 82.65	€ 80.69	€ 73.44	€ 85.05	€ 90.81	€ 83.62	€ 91.53	€ 95.76
	To	€ 61.8	€ 50.2	€ 53.2	€ 48.8	€ 50.0	€ 53.0	€ 55.6	€ 62.2	€ 60.8
	Net Cost of Switch	-€ 44.2	-€ 32.5	-€ 27.5	-€ 24.7	-€ 35.1	-€ 37.8	-€ 28.0	-€ 29.3	-€ 35.0

Table 7-28: Switch from Landfill to Different Biowaste Treatments in the EU27 Using Private Metric

Technology	Cost	AT	BE(Flanders)	BE(Wallonia)	BE(Brussels)	BG	CY	CZ	DK	EE	FI
Landfill to AD: on-site biogas use (elec)	From	€ 114.2	€ 179.7	€ 179.7	€ 179.7	€ 78.8	€ 71.8	€ 113.4	€ 151.2	€ 83.4	€ 122.7
	To	€ 84.5	€ 110.4	€ 110.4	€ 110.4	€ 77.0	€ 61.3	€ 53.0	€ 101.4	€ 66.3	€ 107.7
	Net Cost of Switch	-€ 29.69	-€ 69.23	-€ 69.23	-€ 69.23	-€ 1.77	-€ 10.49	-€ 60.41	-€ 49.77	-€ 17.14	-€ 14.99
Landfill to AD: on-site biogas use + CHP	From	€ 114.15	€ 179.67	€ 179.67	€ 179.67	€ 78.79	€ 71.80	€ 113.41	€ 151.18	€ 83.40	€ 122.69
	To	€ 96.2	€ 124.6	€ 124.6	€ 124.6	€ 89.7	€ 78.1	€ 61.6	€ 115.2	€ 79.5	€ 125.2
	Net Cost of Switch	-€ 18.0	-€ 55.0	-€ 55.0	-€ 55.0	€ 11.0	€ 6.3	-€ 51.8	-€ 35.9	-€ 3.9	€ 2.5
Landfill to AD: compressed biogas used in vehicles	From	€ 114.15	€ 179.67	€ 179.67	€ 179.67	€ 78.79	€ 71.80	€ 113.41	€ 151.18	€ 83.40	€ 122.69
	To	€ 81.9	€ 114.8	€ 114.8	€ 114.8	€ 75.0	€ 98.5	€ 69.5	€ 132.1	€ 93.5	€ 99.2
	Net Cost of Switch	-€ 32.2	-€ 64.9	-€ 64.9	-€ 64.9	-€ 3.8	€ 26.7	-€ 43.9	-€ 19.1	€ 10.1	-€ 23.5
Landfill to AD: biogas injected to gas grid	From	€ 114.15	€ 179.67	€ 179.67	€ 179.67	€ 78.79	€ 71.80	€ 113.41	€ 151.18	€ 83.40	€ 122.69
	To	€ 94.8	€ 108.2	€ 108.2	€ 108.2	€ 83.0	€ 107.3	€ 63.4	€ 121.6	€ 84.5	€ 117.3
	Net Cost of Switch	-€ 19.4	-€ 71.5	-€ 71.5	-€ 71.5	€ 4.2	€ 35.5	-€ 50.0	-€ 29.6	€ 1.1	-€ 5.4
Landfill to IVC	From	€ 114.15	€ 179.67	€ 179.67	€ 179.67	€ 78.79	€ 71.80	€ 113.41	€ 151.18	€ 83.40	€ 122.69
	To	€ 81.5	€ 89.2	€ 89.2	€ 89.2	€ 63.8	€ 69.1	€ 68.1	€ 89.7	€ 66.6	€ 84.7
	Net Cost of Switch	-€ 32.7	-€ 90.4	-€ 90.4	-€ 90.4	-€ 15.0	-€ 2.7	-€ 45.3	-€ 61.5	-€ 16.8	-€ 38.0

Table 7-29: Switch from Landfill to Different Biowaste Treatments in the EU27 Using Private Metric (ctd)

Technology	Cost	FR	DE	EL	HU	IE	IT	LV	LT	LU	MT
Landfill to AD: on-site biogas use (elec)	From	€ 119.7	€ 111.8	€ 81.3	€ 89.3	€ 106.2	€ 112.8	€ 79.6	€ 79.6	€ 114.3	€ 78.2
	To	€ 99.2	€ 69.6	€ 75.1	€ 71.2	€ 90.5	€ 68.5	€ 56.8	€ 78.8	€ 101.5	€ 61.3
	Net Cost of Switch	-€ 20.46	-€ 42.22	-€ 6.14	-€ 18.02	-€ 15.70	-€ 44.35	-€ 22.84	-€ 0.77	-€ 12.79	-€ 16.86
Landfill to AD: on-site biogas use + CHP	From	€ 119.69	€ 111.78	€ 81.29	€ 89.25	€ 106.23	€ 112.83	€ 79.61	€ 79.57	€ 114.32	€ 78.15
	To	€ 113.7	€ 81.3	€ 91.8	€ 82.9	€ 106.9	€ 80.7	€ 69.0	€ 90.8	€ 115.6	€ 77.8
	Net Cost of Switch	-€ 6.0	-€ 30.5	€ 10.5	-€ 6.3	€ 0.7	-€ 32.1	-€ 10.6	€ 11.2	€ 1.3	-€ 0.3
Landfill to AD: compressed biogas used in vehicles	From	€ 119.69	€ 111.78	€ 81.29	€ 89.25	€ 106.23	€ 112.83	€ 79.61	€ 79.57	€ 114.32	€ 78.15
	To	€ 102.2	€ 92.2	€ 98.8	€ 90.8	€ 119.7	€ 94.7	€ 89.4	€ 90.6	€ 114.7	€ 98.4
	Net Cost of Switch	-€ 17.4	-€ 19.6	€ 17.5	€ 1.5	€ 13.5	-€ 18.2	€ 9.8	€ 11.0	€ 0.4	€ 20.2
Landfill to AD: biogas injected to gas grid	From	€ 119.69	€ 111.78	€ 81.29	€ 89.25	€ 106.23	€ 112.83	€ 79.61	€ 79.57	€ 114.32	€ 78.15
	To	€ 105.9	€ 92.1	€ 107.0	€ 78.4	€ 105.1	€ 102.1	€ 77.3	€ 77.7	€ 107.0	€ 105.8
	Net Cost of Switch	-€ 13.8	-€ 19.7	€ 25.7	-€ 10.8	-€ 1.1	-€ 10.7	-€ 2.3	-€ 1.9	-€ 7.4	€ 27.6
Landfill to IVC + biofilter	From	€ 119.69	€ 111.78	€ 81.29	€ 89.25	€ 106.23	€ 112.83	€ 79.61	€ 79.57	€ 114.32	€ 78.15
	To	€ 85.3	€ 82.3	€ 69.1	€ 67.1	€ 82.0	€ 82.6	€ 64.8	€ 65.3	€ 86.3	€ 69.2
	Net Cost of Switch	-€ 34.4	-€ 29.4	-€ 12.2	-€ 22.1	-€ 24.2	-€ 30.2	-€ 14.8	-€ 14.3	-€ 28.0	-€ 9.0

Table 7-30: Switch from Landfill to Different Biowaste Treatments in the EU27 Using Private Metric (ctd)

Technology	Cost	NL	PL	PT	RO	SK	SI	ES	SE	UK
Landfill to AD: on-site biogas use (elec)	From	€ 197.0	€ 87.6	€ 104.9	€ 78.2	€ 97.8	€ 115.0	€ 81.4	€ 135.8	€ 149.2
	To	€ 93.5	€ 59.1	€ 66.5	€ 76.0	€ 74.6	€ 80.1	€ 83.7	€ 104.2	€ 58.0
	Net Cost of Switch	-€ 103.47	-€ 28.48	-€ 38.35	-€ 2.22	-€ 23.28	-€ 34.85	€ 2.32	-€ 31.57	-€ 91.23
Landfill to AD: on-site biogas use + CHP	From	€ 196.97	€ 87.58	€ 104.86	€ 78.22	€ 97.84	€ 114.99	€ 81.39	€ 135.79	€ 149.19
	To	€ 107.3	€ 69.5	€ 83.2	€ 89.3	€ 85.5	€ 91.9	€ 101.1	€ 119.0	€ 74.5
	Net Cost of Switch	-€ 89.7	-€ 18.1	-€ 21.7	€ 11.1	-€ 12.3	-€ 23.1	€ 19.7	-€ 16.8	-€ 74.7
Landfill to AD: compressed biogas used in vehicles	From	€ 196.97	€ 87.58	€ 104.86	€ 78.22	€ 97.84	€ 114.99	€ 81.39	€ 135.79	€ 149.19
	To	€ 108.9	€ 91.1	€ 88.1	€ 90.2	€ 63.0	€ 97.0	€ 92.1	€ 87.1	€ 97.0
	Net Cost of Switch	-€ 88.1	€ 3.5	-€ 16.7	€ 11.9	-€ 34.9	-€ 17.9	€ 10.7	-€ 48.7	-€ 52.2
Landfill to AD: biogas injected to gas grid	From	€ 196.97	€ 87.58	€ 104.86	€ 78.22	€ 97.84	€ 114.99	€ 81.39	€ 135.79	€ 149.19
	To	€ 104.9	€ 70.6	€ 92.3	€ 86.3	€ 69.2	€ 78.0	€ 94.1	€ 110.0	€ 109.4
	Net Cost of Switch	-€ 92.1	-€ 16.9	-€ 12.5	€ 8.1	-€ 28.7	-€ 37.0	€ 12.7	-€ 25.8	-€ 39.7
Landfill to IVC + biofilter	From	€ 196.97	€ 87.58	€ 104.86	€ 78.22	€ 97.84	€ 114.99	€ 81.39	€ 135.79	€ 149.19
	To	€ 87.4	€ 66.6	€ 71.7	€ 64.4	€ 66.6	€ 71.5	€ 74.2	€ 85.4	€ 84.0
	Net Cost of Switch	-€ 109.6	-€ 21.0	-€ 33.2	-€ 13.8	-€ 31.2	-€ 43.5	-€ 7.2	-€ 50.3	-€ 65.2

7.11 Switching from Incineration to Organic Treatment Systems

7.11.1 Cost of Switch

The costs associated with switches are as described in previous sections relating to incineration and IVC/AD technologies. The private costs in **Table 7-31** use discount rates representing the technology specific weighted average cost of capital, and include taxes and subsidies, such as renewable energy support schemes. The social costs **Table 7-28** ignore taxes and subsidies, and use the standard 4% discount rate. The costs for each treatment type show the costs of treating a tonne of biowaste. To calculate the net costs or benefits associated with switching from one treatment route to another, we look at the differences between the treatment costs as outlined in 7.10.3

7.11.2 Environmental Impact of Switch

The external damage costs associated with treating food waste using incineration (shown in Table 7-19 above) are heavily influenced by the energy mix of the country. Damage costs for incineration may be negative (implying a welfare benefit) where the energy mix of the country is based on the more polluting fuels such as coal or oil. This is the case for Malta, Cyprus and Poland, all of which are heavily dependent on oil and coal for the generation of electricity, This effect is exacerbated where the damage costs for SOx are high, such as in the case of Germany and the Netherlands.

Where the external damage cost for incineration is negative, the benefit associated with treating food using AD and using the biogas to generate electricity are typically greater still, due to the more significant air quality impacts associated with the use of incineration in the majority of Member States. In the case of Malta, however, the environmental damage costs associated with injecting the upgraded biogas to grid are more significant than those associated within incineration generating only electricity, as a result of that country's dependence on oil for its electricity generation.

In countries with a more heavily polluting fuel mix, incineration may offer greater environmental benefits than IVC as a result of the greater offset associated with the generation of electricity, since there is always an environmental cost associated with the use of IVC.

However, for the vast majority of Member States with a more typical (or less polluting) fuel mix, the environmental damage costs associated with treating biowaste using any of the AD options or IVC are less significant than those resulting from incineration.

7.11.3 Net Benefit of Switch

The tables below show the social welfare change when a tonne of waste is switched from landfill, to the different biowaste treatments. We assume that the issue of waste growth associated with free garden waste collections, as discussed in Section 7.4.1, are mitigated by maintaining a marginal incentive to not generate additional biowaste (i.e. through maintaining a marginal charge).

Table 7-31 shows this for all countries for the 'private cost' metric, **Table 7-28** for the 'social cost' metric. The 'Net Cost of Switch' figure represents the outcome of the following equation

(RESIDUAL WASTE FINANCIAL COST + RESIDUAL WASTE EXTERNAL COST) *MINUS*

(BIOWASTE FINANCIAL COST + BIOWASTE EXTERNAL COST)

A negative figure implies a reduction in the net financial plus external costs, so a welfare gain.

The best performing switches from incineration for each Member State, and the subsequent ordering of the other switching options, are the same as those outlined in **Table 7-15** where the lowest cost option of the AD variations and IVC are detailed, as the impacts from incineration (for each individual member state) are held constant when compared against each of the organic treatment systems.

A number of factors have a large influence on the extent of the net benefit for individual Member States of moving from incineration to the organic treatment options.

- The costs associated with treatment via incineration are an important factor, so where incineration taxes are high (such as Sweden) the net benefit of switches to organic treatment systems will tend to be higher (under the private metric). Likewise, where the costs associated with incineration are low, for example again in Cyprus, where the revenues from electricity generation are particularly high, switches to organic options, especially where no electricity is generated, provide much lower benefits. In the example of AD gas-to-grid for Cyprus, there is effectively no gas grid on Cyprus, so the revenue is stated as zero.
- The costs of incineration are particularly important when considering the switch to IVC, as there is little variation in IVC costs between member states. The only source of variation within the financial costs for IVC is the cost of labour, as there are no revenues from energy generation or supporting fiscal mechanisms
- The variation in external benefits from switching to particular organic treatment systems is dependant on the energy mix and the level of the damage costs attributed to each tonne of pollutant for each country, as was described in the environmental cost sections for the different technologies. Countries that are reliant on coal or oil for their electricity supply (such as the Czech Republic) will typically see a greater level of benefit from incineration as this is offsetting a greater amount of GHG and non-GHG emissions. These benefits may, however, be reduced where the country concerned has relatively low damage costs per tonne of pollution.
- Where electricity generation has a lower carbon intensity, such as in Sweden, the benefits of producing electricity from AD are reduced relative to the production of biogas for vehicle fuel, by comparison with other countries where electricity generation with a higher carbon intensity. Looking ahead, with various policy drivers working to reduce the carbon intensity of electricity generation, it may well be that the use of biogas to replace vehicle fuel becomes increasingly attractive.

It is important to note that almost all of the switches from incineration provide a net social benefit. This highlights the importance of diverting food waste from incineration. The two notable exceptions to this are:

- Cyprus, where the switches to AD gas-to-grid, and AD for vehicle fuel show net costs under both metrics. This is effectively because there is no revenue assumed for gas to the grid, or for use as vehicle fuel. Conversely the price received for electricity sales is high.[88]

- Malta, where the high price received for electricity provides an additional financial benefit associated with the generation of electricity from incineration. In addition, the country has no gas grid, and thus no revenue from gas sales. The value for biogas is also very low. These factors result in a net cost for those biogas switches under the social metric.

7.11.3.1 The Issue of Costs

It is very important to place these figures into context with the discussion regarding the costs of the collection logistics discussed above. We concentrate here on two key points:

- The costs of the collection service; and
- The potential effects on waste prevention and recycling.

7.11.3.1.1 Collection Costs

In very few countries are the benefits of switching from landfill or incineration to the preferred biowaste treatment option less than around €30 per tonne. In principle, since we are discussing the switching of food waste from one treatment system to another, and since, in the various different countries of the EU, we might expect that a well operated system for food waste will capture something of the order 65kg per capita, then implications of this outcome might suggest that as long as the collection system itself does not increase costs more than around €2 per capita, then a net benefit would still accrue to society from the switches discussed.

In this respect, the issue of whether these benefits will be secured for society does indeed come down to the degree to which collection systems are optimised, particularly in respect of collection frequencies, vehicle choice and container choice, to deliver a well-functioning system at efficient cost.

7.11.3.1.2 Potential effects on waste prevention and recycling

The assumption in many studies of this nature is that the systems can be compared on an equivalent 'tonne for tonne' basis. The presumption is that the collection system itself has no effect on the mass flows within the waste system. We know this not to be true (see Section 7.4.1.1). The collection system itself affects the way in which people behave, and there are two effects which deserve to be mentioned in this regard precisely because they are of such significance:

The first is the effect of the provision of targeted kitchen waste collections upon household food waste generation. As discussed above, there are likely to be significant benefits associated with the prevention of food waste. These have been estimated as being as high as 4.5 tonnes of CO_2 per tonne of food waste.

Households in the EU may produce as much as 200kg per household per annum. One recent UK case which has been well observed suggests a prevention effect as high as 25% may be associated with the collection system. This would imply that for every tonne of food waste being landfilled or incinerated, the appropriate comparator might be 0.75 tonnes of food being treated and 0.25 tonnes prevented. The 0.25 tonnes prevented, if evaluated at a level of 4.5 tonnes CO_2 saved per tonne prevented, would give a net

[88] Even without undertaking this analysis, one would most likely assume that these options were not going to be reasonable ones to undertake on Cyprus. As a Mediterranean country, the demand for heat is low, and with no gas grid, it is unreasonable to expect injection of biogas to take place

saving of 1.125 tonnes CO_2 per tonne of food waste considered. In the comparative analysis, therefore, there would be a considerable additional benefit, whilst the introduction of the system would also affect treatment costs.

Figure 7-21: Net Cost to Society from Switch of Biowaste from Incineration / Landfill to Lowest Cost Option (Social Cost Metric)

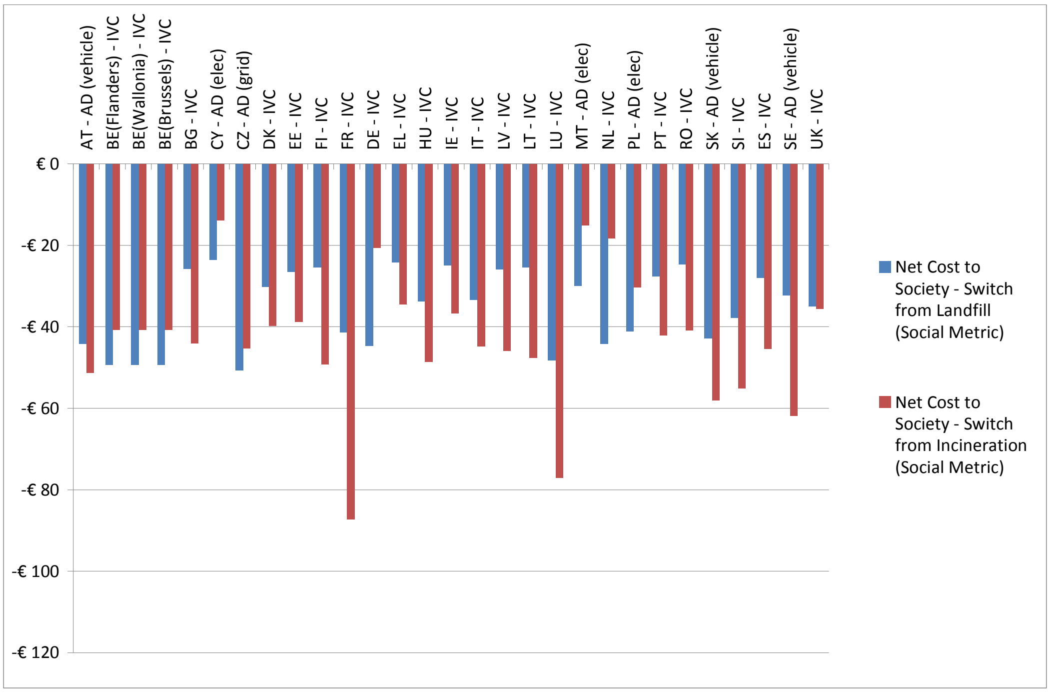


Figure 7-22: Net Cost to Society from Switch of Biowaste from Incineration / Landfill to Lowest Cost Option (Private Cost Metric)

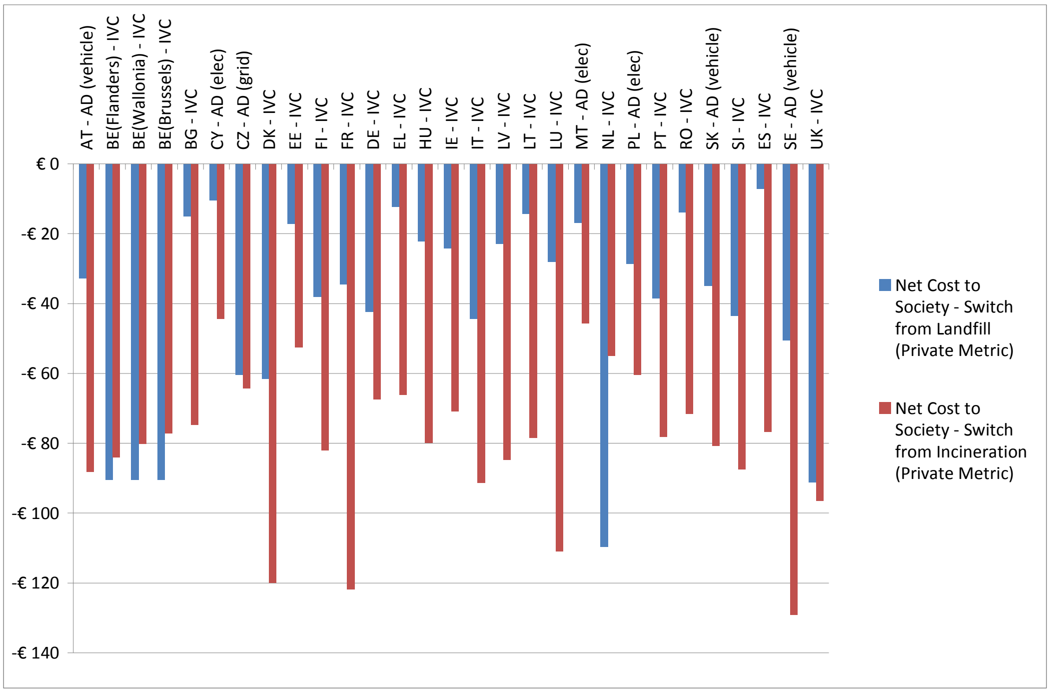


Table 7-31: Switch from Incineration to Different Biowaste Treatments in the EU27 Using Social Metric

Technology	Cost	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI
Incineration to AD: on-site biogas use (elec)	From	€ 107.5	€ 104.8	€ 104.8	€ 104.8	€ 92.3	€ 57.0	€ 87.1	€ 104.5	€ 88.7	€ 111.3
	To	€ 80.7	€ 84.4	€ 84.4	€ 84.4	€ 60.4	€ 43.2	€ 43.8	€ 75.6	€ 59.0	€ 85.3
	Net Cost of Switch	-€ 26.77	-€ 20.42	-€ 20.42	-€ 20.42	-€ 31.90	-€ 13.73	-€ 43.35	-€ 28.91	-€ 29.74	-€ 25.96
Incineration to AD: on-site biogas use + CHP	From	€ 107.51	€ 104.85	€ 104.85	€ 104.85	€ 92.35	€ 56.97	€ 87.13	€ 104.53	€ 88.72	€ 111.26
	To	€ 86.6	€ 92.4	€ 92.4	€ 92.4	€ 68.7	€ 55.1	€ 47.6	€ 83.4	€ 67.5	€ 96.8
	Net Cost of Switch	-€ 20.9	-€ 12.4	-€ 12.4	-€ 12.4	-€ 23.7	-€ 1.8	-€ 39.5	-€ 21.1	-€ 21.2	-€ 14.5
Incineration to AD: compressed biogas used in vehicles	From	€ 107.51	€ 104.85	€ 104.85	€ 104.85	€ 92.35	€ 56.97	€ 87.13	€ 104.53	€ 88.72	€ 111.26
	To	€ 56.2	€ 84.8	€ 84.8	€ 84.8	€ 55.6	€ 77.3	€ 48.0	€ 102.4	€ 73.2	€ 71.9
	Net Cost of Switch	-€ 51.3	-€ 20.0	-€ 20.0	-€ 20.0	-€ 36.8	€ 20.4	-€ 39.2	-€ 2.2	-€ 15.5	-€ 39.3
Incineration to AD: biogas injected to gas grid	From	€ 107.51	€ 104.85	€ 104.85	€ 104.85	€ 92.35	€ 56.97	€ 87.13	€ 104.53	€ 88.72	€ 111.26
	To	€ 69.0	€ 78.3	€ 78.3	€ 78.3	€ 63.6	€ 86.1	€ 41.9	€ 91.8	€ 64.2	€ 90.0
	Net Cost of Switch	-€ 38.5	-€ 26.6	-€ 26.6	-€ 26.6	-€ 28.7	€ 29.1	-€ 45.3	-€ 12.7	-€ 24.5	-€ 21.2
Incineration to IVC + biofilter	From	€ 107.51	€ 104.85	€ 104.85	€ 104.85	€ 92.35	€ 56.97	€ 87.13	€ 104.53	€ 88.72	€ 111.26
	To	€ 60.4	€ 64.2	€ 64.2	€ 64.2	€ 48.3	€ 52.1	€ 50.7	€ 64.9	€ 50.0	€ 62.1
	Net Cost of Switch	-€ 47.1	-€ 40.7	-€ 40.7	-€ 40.7	-€ 44.0	-€ 4.9	-€ 36.5	-€ 39.7	-€ 38.7	-€ 49.1

Table 7-32: Switch from Incineration to Different Biowaste Treatments in the EU27 Using Social Metric (ctd)

Technology	Cost	FR	DE	EL	HU	IE	IT	LV	LT	LU	MT
Incineration to AD: on-site biogas use (elec)	From	€ 150.3	€ 81.8	€ 86.6	€ 99.2	€ 97.3	€ 105.3	€ 94.8	€ 96.9	€ 141.2	€ 58.2
	To	€ 93.6	€ 66.0	€ 60.4	€ 56.4	€ 68.2	€ 68.3	€ 61.2	€ 61.8	€ 85.8	€ 43.2
	Net Cost of Switch	-€ 56.73	-€ 15.84	-€ 26.25	-€ 42.77	-€ 29.16	-€ 37.00	-€ 33.59	-€ 35.11	-€ 55.37	-€ 14.99
Incineration to AD: on-site biogas use + CHP	From	€ 150.33	€ 81.83	€ 86.61	€ 99.16	€ 97.32	€ 105.31	€ 94.77	€ 96.91	€ 141.22	€ 58.21
	To	€ 101.9	€ 71.8	€ 72.1	€ 63.3	€ 78.7	€ 74.7	€ 68.8	€ 69.2	€ 93.7	€ 54.9
	Net Cost of Switch	-€ 48.4	-€ 10.1	-€ 14.5	-€ 35.8	-€ 18.6	-€ 30.6	-€ 25.9	-€ 27.7	-€ 47.6	-€ 3.4
Incineration to AD: compressed biogas used in vehicles	From	€ 150.33	€ 81.83	€ 86.61	€ 99.16	€ 97.32	€ 105.31	€ 94.77	€ 96.91	€ 141.22	€ 58.21
	To	€ 75.2	€ 66.3	€ 77.6	€ 70.2	€ 93.6	€ 68.7	€ 69.7	€ 70.7	€ 87.9	€ 77.2
	Net Cost of Switch	-€ 75.1	-€ 15.5	-€ 9.0	-€ 29.0	-€ 3.7	-€ 36.6	-€ 25.1	-€ 26.2	-€ 53.4	€ 19.0
Incineration to AD: biogas injected to gas grid	From	€ 150.33	€ 81.83	€ 86.61	€ 99.16	€ 97.32	€ 105.31	€ 94.77	€ 96.91	€ 141.22	€ 58.21
	To	€ 78.9	€ 66.2	€ 85.9	€ 57.8	€ 79.0	€ 76.1	€ 57.6	€ 57.8	€ 80.2	€ 84.6
	Net Cost of Switch	-€ 71.5	-€ 15.6	-€ 0.8	-€ 41.3	-€ 18.3	-€ 29.2	-€ 37.2	-€ 39.1	-€ 61.1	€ 26.4
Incineration to IVC + biofilter	From	€ 150.33	€ 81.83	€ 86.61	€ 99.16	€ 97.32	€ 105.31	€ 94.77	€ 96.91	€ 141.22	€ 58.21
	To	€ 63.1	€ 61.2	€ 52.1	€ 50.6	€ 60.6	€ 60.7	€ 49.0	€ 49.4	€ 64.2	€ 52.1
	Net Cost of Switch	-€ 87.2	-€ 20.6	-€ 34.5	-€ 48.5	-€ 36.7	-€ 44.7	-€ 45.8	-€ 47.6	-€ 77.0	-€ 6.1

Table 7-33: Switch from Incineration to Different Biowaste Treatments in the EU27 Using Social Metric (ctd)

Technology	Cost	NL	PL	PT	RO	SK	SI	ES	SE	UK
Incineration to AD: on-site biogas use (elec)	From	€ 80.0	€ 71.9	€ 95.2	€ 89.5	€ 100.3	€ 108.0	€ 101.1	€ 121.0	€ 96.4
	To	€ 67.0	€ 41.6	€ 64.6	€ 59.3	€ 56.9	€ 60.6	€ 67.7	€ 88.2	€ 66.7
	Net Cost of Switch	-€ 13.03	-€ 30.25	-€ 30.56	-€ 30.29	-€ 43.47	-€ 47.38	-€ 33.37	-€ 32.82	-€ 29.73
Incineration to AD: on-site biogas use + CHP	From	€ 80.01	€ 71.90	€ 95.21	€ 89.54	€ 100.34	€ 107.97	€ 101.06	€ 121.04	€ 96.43
	To	€ 74.8	€ 47.3	€ 76.3	€ 68.0	€ 63.2	€ 67.3	€ 79.8	€ 97.0	€ 77.4
	Net Cost of Switch	-€ 5.3	-€ 24.6	-€ 18.9	-€ 21.5	-€ 37.2	-€ 40.7	-€ 21.2	-€ 24.0	-€ 19.0
Incineration to AD: compressed biogas used in vehicles	From	€ 80.01	€ 71.90	€ 95.21	€ 89.54	€ 100.34	€ 107.97	€ 101.06	€ 121.04	€ 96.43
	To	€ 78.5	€ 70.7	€ 65.4	€ 70.5	€ 42.3	€ 74.3	€ 69.6	€ 59.3	€ 69.3
	Net Cost of Switch	-€ 1.5	-€ 1.2	-€ 29.8	-€ 19.0	-€ 58.0	-€ 33.7	-€ 31.5	-€ 61.7	-€ 27.2
Incineration to AD: biogas injected to gas grid	From	€ 80.01	€ 71.90	€ 95.21	€ 89.54	€ 100.34	€ 107.97	€ 101.06	€ 121.04	€ 96.43
	To	€ 74.6	€ 50.2	€ 69.7	€ 66.7	€ 48.5	€ 55.2	€ 71.5	€ 82.3	€ 81.7
	Net Cost of Switch	-€ 5.5	-€ 21.7	-€ 25.5	-€ 22.8	-€ 51.8	-€ 52.7	-€ 29.5	-€ 38.7	-€ 14.7
Incineration to IVC + biofilter	From	€ 80.01	€ 71.90	€ 95.21	€ 89.54	€ 100.34	€ 107.97	€ 101.06	€ 121.04	€ 96.43
	To	€ 61.8	€ 50.2	€ 53.2	€ 48.8	€ 50.0	€ 53.0	€ 55.6	€ 62.2	€ 60.8
	Net Cost of Switch	-€ 18.2	-€ 21.7	-€ 42.0	-€ 40.8	-€ 50.4	-€ 55.0	-€ 45.4	-€ 58.8	-€ 35.6

Table 7-34: Switch from Incineration to Different Biowaste Treatments in the EU27 Using Private Metric

Technology	Cost	AT	BE (Flanders)	BE (Wallonia)	BE (Brussels)	BG	CY	CZ	DK	EE	FI
Incineration to AD: on-site biogas use (elec)	From	€ 169.5	€ 173.3	€ 169.3	€ 166.3	€ 138.5	€ 105.6	€ 117.3	€ 209.6	€ 118.8	€ 166.8
	To	€ 84.5	€ 110.4	€ 110.4	€ 110.4	€ 77.0	€ 61.3	€ 53.0	€ 101.4	€ 66.3	€ 107.7
	Net Cost of Switch	-€ 85.06	-€ 62.88	-€ 58.88	-€ 55.88	-€ 61.50	-€ 44.26	-€ 64.28	-€ 108.22	-€ 52.54	-€ 59.05
Incineration to AD: on-site biogas use + CHP	From	€ 169.52	€ 173.31	€ 169.31	€ 166.31	€ 138.52	€ 105.57	€ 117.29	€ 209.62	€ 118.80	€ 166.76
	To	€ 96.2	€ 124.6	€ 124.6	€ 124.6	€ 89.7	€ 78.1	€ 61.6	€ 115.2	€ 79.5	€ 125.2
	Net Cost of Switch	-€ 73.3	-€ 48.7	-€ 44.7	-€ 41.7	-€ 48.8	-€ 27.5	-€ 55.6	-€ 94.4	-€ 39.3	-€ 41.6
Incineration to AD: compressed biogas used in vehicles	From	€ 169.52	€ 173.31	€ 169.31	€ 166.31	€ 138.52	€ 105.57	€ 117.29	€ 209.62	€ 118.80	€ 166.76
	To	€ 81.9	€ 114.8	€ 114.8	€ 114.8	€ 75.0	€ 98.5	€ 69.5	€ 132.1	€ 93.5	€ 99.2
	Net Cost of Switch	-€ 87.6	-€ 58.5	-€ 54.5	-€ 51.5	-€ 63.6	-€ 7.1	-€ 47.8	-€ 77.5	-€ 25.3	-€ 67.5
Incineration to AD: biogas injected to gas grid	From	€ 169.52	€ 173.31	€ 169.31	€ 166.31	€ 138.52	€ 105.57	€ 117.29	€ 209.62	€ 118.80	€ 166.76
	To	€ 94.8	€ 108.2	€ 108.2	€ 108.2	€ 83.0	€ 107.3	€ 63.4	€ 121.6	€ 84.5	€ 117.3
	Net Cost of Switch	-€ 74.8	-€ 65.1	-€ 61.1	-€ 58.1	-€ 55.5	€ 1.7	-€ 53.9	-€ 88.0	-€ 34.3	-€ 49.4
Incineration to IVC + biofilter	From	€ 169.52	€ 173.31	€ 169.31	€ 166.31	€ 138.52	€ 105.57	€ 117.29	€ 209.62	€ 118.80	€ 166.76
	To	€ 81.5	€ 89.2	€ 89.2	€ 89.2	€ 63.8	€ 69.1	€ 68.1	€ 89.7	€ 66.6	€ 84.7
	Net Cost of Switch	-€ 88.0	-€ 84.1	-€ 80.1	-€ 77.1	-€ 74.7	-€ 36.5	-€ 49.2	-€ 119.9	-€ 52.2	-€ 82.1

Table 7-35: Switch from Incineration to Different Biowaste Treatments in the EU27 Using Private Metric (ctd)

Technology	Cost	FR	DE	EL	HU	IE	IT	LV	LT	LU	MT
Incineration to AD: on-site biogas use (elec)	From	€ 207.1	€ 136.9	€ 135.2	€ 147.0	€ 152.9	€ 159.8	€ 141.4	€ 143.8	€ 197.3	€ 106.8
	To	€ 99.2	€ 69.6	€ 75.1	€ 71.2	€ 90.5	€ 68.5	€ 56.8	€ 78.8	€ 101.5	€ 61.3
	Net Cost of Switch	-€ 107.90	-€ 67.30	-€ 60.06	-€ 75.73	-€ 62.35	-€ 91.30	-€ 84.62	-€ 64.99	-€ 95.74	-€ 45.51
Incineration to AD: on-site biogas use + CHP	From	€ 207.13	€ 136.87	€ 135.21	€ 146.97	€ 152.88	€ 159.79	€ 141.38	€ 143.80	€ 197.27	€ 106.81
	To	€ 113.7	€ 81.3	€ 91.8	€ 82.9	€ 106.9	€ 80.7	€ 69.0	€ 90.8	€ 115.6	€ 77.8
	Net Cost of Switch	-€ 93.5	-€ 55.5	-€ 43.4	-€ 64.0	-€ 46.0	-€ 79.1	-€ 72.4	-€ 53.0	-€ 81.7	-€ 29.0
Incineration to AD: compressed biogas used in vehicles	From	€ 207.13	€ 136.87	€ 135.21	€ 146.97	€ 152.88	€ 159.79	€ 141.38	€ 143.80	€ 197.27	€ 106.81
	To	€ 102.2	€ 92.2	€ 98.8	€ 90.8	€ 119.7	€ 94.7	€ 89.4	€ 90.6	€ 114.7	€ 98.4
	Net Cost of Switch	-€ 104.9	-€ 44.7	-€ 36.4	-€ 56.2	-€ 33.2	-€ 65.1	-€ 52.0	-€ 53.2	-€ 82.6	-€ 8.4
Incineration to AD: biogas injected to gas grid	From	€ 207.13	€ 136.87	€ 135.21	€ 146.97	€ 152.88	€ 159.79	€ 141.38	€ 143.80	€ 197.27	€ 106.81
	To	€ 105.9	€ 92.1	€ 107.0	€ 78.4	€ 105.1	€ 102.1	€ 77.3	€ 77.7	€ 107.0	€ 105.8
	Net Cost of Switch	-€ 101.3	-€ 44.8	-€ 28.2	-€ 68.6	-€ 47.8	-€ 57.7	-€ 64.1	-€ 66.1	-€ 90.3	-€ 1.0
Incineration to IVC + biofilter	From	€ 207.13	€ 136.87	€ 135.21	€ 146.97	€ 152.88	€ 159.79	€ 141.38	€ 143.80	€ 197.27	€ 106.81
	To	€ 85.3	€ 82.3	€ 69.1	€ 67.1	€ 82.0	€ 82.6	€ 64.8	€ 65.3	€ 86.3	€ 69.2
	Net Cost of Switch	-€ 121.8	-€ 54.5	-€ 66.1	-€ 79.8	-€ 70.9	-€ 77.2	-€ 76.6	-€ 78.5	-€ 110.9	-€ 37.7

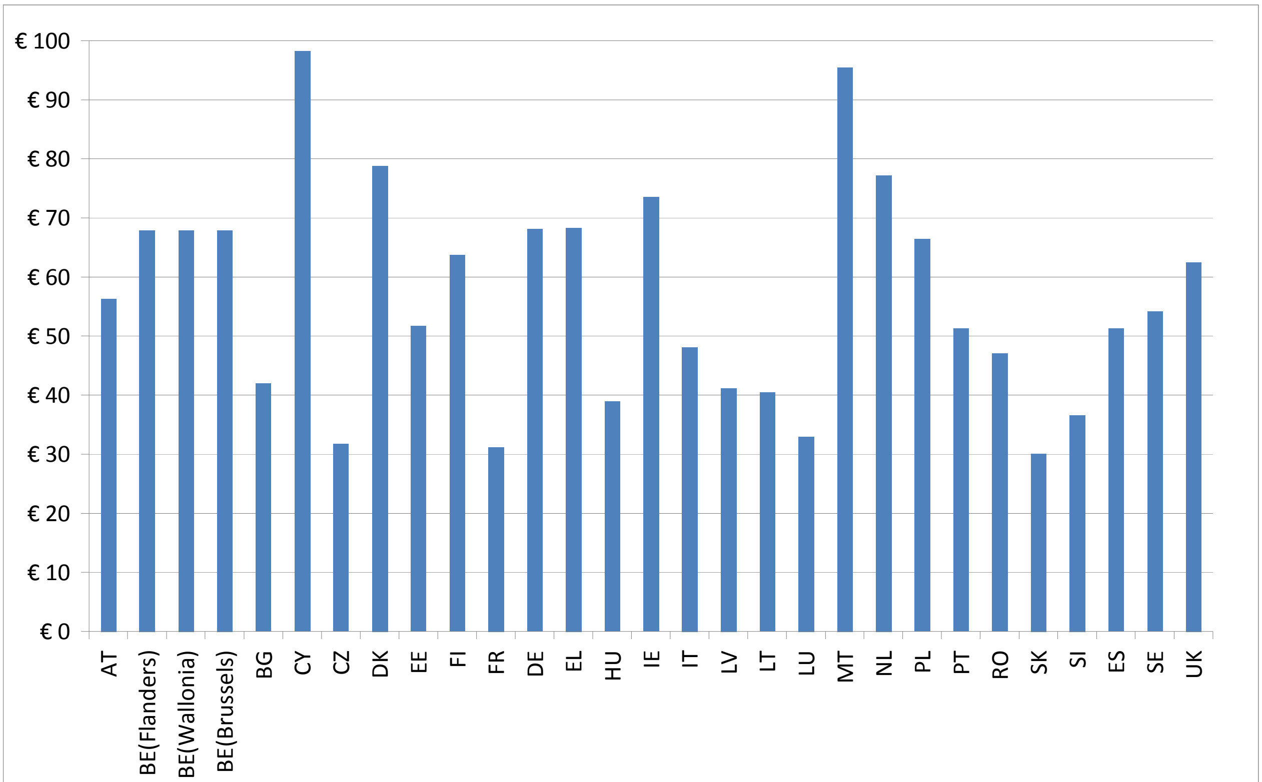
Table 7-36: Switch from Incineration to Different Biowaste Treatments in the EU27 Using Private Metric (ctd)

Technology	Cost	NL	PL	PT	RO	SK	SI	ES	SE	UK
Incineration to AD: on-site biogas use (elec)	From	€ 142.3	€ 119.5	€ 144.6	€ 136.0	€ 143.7	€ 159.0	€ 150.9	€ 214.5	€ 154.5
	To	€ 93.5	€ 59.1	€ 66.5	€ 76.0	€ 74.6	€ 80.1	€ 83.7	€ 104.2	€ 58.0
	Net Cost of Switch	-€ 48.83	-€ 60.40	-€ 78.08	-€ 60.00	-€ 69.17	-€ 78.84	-€ 67.21	-€ 110.25	-€ 96.52
Incineration to AD: on-site biogas use + CHP	From	€ 142.34	€ 119.50	€ 144.59	€ 135.99	€ 143.73	€ 158.98	€ 150.93	€ 214.47	€ 154.48
	To	€ 107.3	€ 69.5	€ 83.2	€ 89.3	€ 85.5	€ 91.9	€ 101.1	€ 119.0	€ 74.5
	Net Cost of Switch	-€ 35.0	-€ 50.0	-€ 61.4	-€ 46.7	-€ 58.2	-€ 67.1	-€ 49.8	-€ 95.5	-€ 79.9
Incineration to AD: compressed biogas used in vehicles	From	€ 142.34	€ 119.50	€ 144.59	€ 135.99	€ 143.73	€ 158.98	€ 150.93	€ 214.47	€ 154.48
	To	€ 108.9	€ 91.1	€ 88.1	€ 90.2	€ 63.0	€ 97.0	€ 92.1	€ 87.1	€ 97.0
	Net Cost of Switch	-€ 33.5	-€ 28.4	-€ 56.5	-€ 45.8	-€ 80.7	-€ 61.9	-€ 58.8	-€ 127.4	-€ 57.5
Incineration to AD: biogas injected to gas grid	From	€ 142.34	€ 119.50	€ 144.59	€ 135.99	€ 143.73	€ 158.98	€ 150.93	€ 214.47	€ 154.48
	To	€ 104.9	€ 70.6	€ 92.3	€ 86.3	€ 69.2	€ 78.0	€ 94.1	€ 110.0	€ 109.4
	Net Cost of Switch	-€ 37.5	-€ 48.9	-€ 52.2	-€ 49.7	-€ 74.5	-€ 81.0	-€ 56.8	-€ 104.4	-€ 45.0
Incineration to IVC + biofilter	From	€ 142.34	€ 119.50	€ 144.59	€ 135.99	€ 143.73	€ 158.98	€ 150.93	€ 214.47	€ 154.48
	To	€ 87.4	€ 66.6	€ 71.7	€ 64.4	€ 66.6	€ 71.5	€ 74.2	€ 85.4	€ 84.0
	Net Cost of Switch	-€ 55.0	-€ 52.9	-€ 72.9	-€ 71.6	-€ 77.1	-€ 87.4	-€ 76.7	-€ 129.0	-€ 70.5

7.11.4 Sensitivity Around Existing Incineration Capacity

Some Member States have substantial existing incineration capacity. Where low quantities of biowaste are being landfilled, the net cost to society of reducing the need for this capacity and installing of alternative infrastructure, may be significantly different to the switch costs represented in the preceding Section. If policies imposed a requirement to effectively substitute new biowaste treatment capacity for incineration, then it might be argued that since the capital element has already been 'sunk', the modelling of the financial costs of the switch, and the net cost to society, would need to take into account the sunk capital costs. Figure 7-23, below, shows what happens when the capital costs of the existing infrastructure are excluded from the calculation of the benefits of the switch to biowaste treatments. What this shows is that there is a net *cost* to society in closing down existing incineration capacity to build new biowaste treatment facilities where the capital cannot be put to any useful alternative purpose. However, as Section 7.11 shows, at the margin, it is more beneficial to switch biowaste to source segregated facilities.

Figure 7-23: Net Cost to Society of Switch from Incineration (with no Capital Cost) to Biowaste Treatment, social metric



It is important to note that this very much represents a 'worst case' scenario. A key point to understand in interpreting this sensitivity analysis is the way the modelling in this study is configured, in terms of 'switching' from one treatment process to another. In the policy scenarios modelled, the 'switch' from incineration to biowaste treatment does not literally represent the switching of waste away from an existing incineration facility to a 'replacement' biowaste treatment facility. Rather, it serves to illustrate the effects of decisions made under different policy scenarios, these being made over an extended period of time. The countries most likely to be affected by this type of situation, therefore, are those that already have significant incineration capacity in place (or have advanced plans of this nature).

In reality, a number of additional factors would serve to reduce the extent to which the issue of sunk costs is genuinely applicable:

- The problem of switching from incineration to other treatments is only a potential problem for capacity that already exists. Since the policies proposed take effect over an extended period of time, it would be expected that developers would adjust their decisions in respect of any investment in additional incineration capacity or in upgrades / retrofits to facilities which are reaching the end of their useful life;
- This would then mean, assuming a lifetime of 20 years for incinerators, that some existing incinerators could be phased out over the time during which the policy took effect, or otherwise, that replacement facilities could be sized accordingly. Therefore the above situation, in which the extreme case of a zero cost for capital was implied, would most likely be associated only with capacity already, or soon to be, in place, and then, the problem would be reduced for existing facilities owing to the time over which the policy would take effect. On average, the worst case might (see below) apply to on average, half the level of the diversion of biowaste from existing incinerators (though the actual figure would depend upon the age profile of incineration capacity in the respective Member State).

Notwithstanding the above, it is also possible that municipal waste incinerators would be able to seek alternative input material, from both commercial and industrial sources, and, if classified as an R1 recovery installation, potentially from other countries within the EU. Obviously, in this last case, the net revenue received by the facility for treating waste might be lower (owing to transport costs).

It should be noted that at the EU level, even in the baseline in 2020, many nations are expected to be still sending considerable quantities of material direct to landfill. However, it should also be noted that some countries already show some tendency towards surplus capacity for their needs, even without the anticipated policies.

As an example of the scale of the impacts, Scenario 3 indicates that in 2020, there will be a reduction in treatment via incineration, compared to the baseline, of 2.5 million tonnes. This equates to around thirteen fewer 200,000 tonnes per annum facilities across the whole of Europe, before accounting for any of the mitigating factors such as advanced notice of policy implementation as outlined above.

The change relative to Scenario 2 is partly related to waste prevention. The above analysis (in Figure 7-23) does not hold in the case of waste prevention. In this case, even assuming no capital cost associated with incineration, there is a net social benefit. As such, with regard to this prevention effect, it could be argued that one should not be overly concerned with the implications for incineration capacity and sunk costs. Indeed, to be concerned for this would effectively be to undermine not only the waste hierarchy, but the suggested primacy of waste prevention in this instance.

Finally, in respect of incineration, it is worth considering in a little more detail the nature of the materials being captured under the Scenarios, and the possible implications for incinerators more generally. Concerning food waste, depending on moisture content as received, the net calorific of the waste can be quite low (perhaps 3-5 MJ/kg, as compared with 9-10 MJ/kg for mixed residual waste). As such, reducing the content of food waste in residual waste will tend to increase its average calorific value. Since most incinerators are limited by the thermal content of the waste they combust, the effect of removing greater quantities of food waste from the waste stream would most likely raise net calorific value of residual waste (other things being equal), thereby reducing the overall quantity of such waste which could be handled by such facilities. Consequently, the effect of removing a given quantity of biowaste from incineration might exaggerate the effect on the incinerator in terms of capacity (because this is essentially limited by the calorific content of the waste being dealt with). On the other hand, to the extent that the financial viability of waste incinerators rests upon charging a gate fee for each tonne of waste, other things being equal, the tendency might be to increase incinerator gate fees at the margin since a given facility would be capable of dealing with a lower quantity of waste with a higher calorific value. Similar considerations apply with garden waste, though the moisture content tends to be lower, and calorific values tend to be slightly higher.

It should be noted that this issue of stranded assets is likely to be somewhat less worrying for countries where the principle effect on residual waste is to reduce the required capacity at MBT plants. One reason for supposing this to be the case is that the nature of MBT is likely, in most cases, to offer some opportunity to reconfigure facilities for processing of source segregated biowaste. Some of the technical components of the plant may be made more or less redundant, but much of the facility may be usefully applied to dealing with source segregated materials. The effect will ultimately be specific to the design of the plant.

7.12 Estimate of employment effects

According to RPA[89], statistical data collected at the European level provides poor quality information on waste management-related employment because:

- the classifications used exclude a wide range of waste related activities; and
- few countries submit regular, up-to-date information.

Using the results of specialised studies (country-, waste stream-, or activity-specific) , RPA have concluded that the probable *level* of employment in the EU15 in organisations for which waste management is a *primary* activity totals around 200,000 to 400,000 (approximately 0.2-0.4% of total EU15 employment in 2001).

However, what we are interested here are changes in employment at the EU27 level following changes in biowaste management – even less data exists that could be used to estimate these changes.

RPA for instance has concluded that:

- The most labour intensive activities (< 500 tonnes of waste per job) are manual sorting, some separate collection processes and waste and scrap wholesale.
- The least labour intensive activities (> 500 tonnes of waste per job) are landfill, incineration and composting together with most forms of collection.

Of course, this is not precise enough for the purposes of our study.

One study undertaken for the European Commission[90] gives the following estimate of the number of employees that are needed per 100 000 tons in composting plants:

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[89] RPA Ltd (2001), Employment Effects of Waste Management Policies, final report prepared for the European Commission, DG ENV

[90] Le Bozec (2004), Costs models for each municipal solid waste process, deliverable 5 and 7 of AWAST, project funded by the 5th FP.

Installation type	Total employees	Technical staff	Administrative staff
Turned windrow composting	31		
Rotating drum IVC	25	22	3
Enclosed hall IVC	51	41	1

Thus, even for IVC, there is a factor 2 difference between the most and the least labour intensive waste management option – this indicates that the actual labour intensity of a composting plant is highly dependent on the technology adopted. The AWAST study has not provided similar estimates for the other waste management options that we consider here.

Another, more recent survey of waste technologies, gives the following estimates[91]:

- MBT: Staffing levels (including technical competence, management and administrative resources) will vary depending on the size and technology adopted;
- Waste incineration: An estimated 30-55 persons are required to operate medium sized facilities of around 200 000 – 450 000 tonnes per annum;
- AD: Staffing levels (including technical competence, management and administrative resources) will vary depending on the size of the facility;
- Composting (both windrow and IVC): Staffing levels (including technical competence, management and administrative resources) will vary depending on the size and technology adopted;

Thus, while incinerators are generally less labour intensive than composting, depending on the technology and the size of the facility, there is an overlap in the estimates of labour intensity. For MBT and AD, even orders of magnitude are not reported.

For indicative purposes, we shall assume that the estimates provided by Last and Le Bozec are comparable.

The figures above indicate that incinerators need at the minimum 6 persons to treat 100 000 tonnes of biowaste, and composting plants at the most 51. If we assume 0.4 tonnes of biowaste per household, the difference in employment related to the two waste management options is 176 people per million households. Alternatively, if we compare the central values of the estimates (17 persons in the case of incinerators and 37.5 persons in the case of composting plants), the difference in employment per million households is 82 people.

We are not aware of recent estimate of the number of households in the EU27.

In a 2001 study[92], the European Environment Agency has estimated the average size of a household in the EEA to be 2.5 persons. Eurostat reports the population of the EU27 to be approximately 500 million people. If we assume that the data related to the EEA can be extrapolated to the EU27, this would then correspond to 200 million households.

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[91] Steve Last, An Introduction to Waste Technologies, January 2008.

[92] http://themes.eea.europa.eu/Sectors_and_activities/households/indicators/consumption/hh03householdnumbersize.pdf

In the baseline, it is project that by 2020, 23 million tonnes of biowaste will be incinerated, out of a total of 97 million tonnes, or 24%.

For purely illustrative purposes, this means that if, in 2020, we would compost all biowaste that is projected to be incinerated, this would lead to a direct job creation effect of 3936 (= 0.24*82*200) to 8 488 (= 0.24*176*200) units.

A similar thought experiment can by undertaken for changes in the waste collection system.

For illustrative purposes, let us compare 2 scenarios.

	Scenario 1	Scenario 2
Organic collection system 1	Fortnightly. Driver plus two operatives. Food and garden waste	Fortnightly. Driver plus two operatives. Garden waste only
Daily pass rate	1400	1500
Staff per million households	**214**	**200**
Organic collection system 2	None	Fortnightly. Driver plus an average 0,75 operatives. Food waste
Daily pass rate		900
Staff per million households		**389**
Residual waste collection	Weekly driver plus 2	Fortnightly driver plus 2
Daily pass rate	1400	1500
Staff per million households	**429**	**200**
Total employed per million households	**643**	**789**
Difference		**146**

In this table, “daily pass rate” refers to the number of households a vehicle will serve (or pass) in any one working day. I.e., if you have 10,000 households in your authority and a vehicle can serve 1,000 per day, on a weekly collection system you need two vehicles.

The changes between the two scenarios are:

- In scenario 1, there is a fortnightly collection of food and garden waste.
- In scenario 2, there is a fortnightly collection of garden waste, and a weekly collection of food waste.
- Residual waste collection frequency can drop from a weekly to a fortnightly collection.
- In the 2nd scenario, the daily pass rate increased from 1400 to 1500 higher because there is less material to collect.

Thus, the difference in collection system yields an estimated difference in employment related to collection activities of 146 units per million household.

Using the same data as above, this yields an estimate of 29200 (= 146*200) new jobs at the EU27 if the EU27 would move *entirely* from biowaste collection scenario A to B. Of course, this figure is again purely indicative.

To summarize this discussion: a change in waste management options as described above could lead to the *direct* creation of a few thousands jobs at the EU27 level in composting activities and maybe a few tens of thousands jobs in waste collection.

However, we think this conclusion should be interpreted very cautiously.

First, the policy scenarios we investigate cover a much wider variety of switches than just from incineration to composting. However, it is not possible to give reasonable estimates of the direct employment effects linked to the other switches.

Second, no detailed information at the national level is available concerning current collection practices.

Third, and more fundamentally, the fact that one waste management option is more labour intensive than the other does not prove that this leads to a "net" job creation in the economy as a whole. Somebody has to pay for this higher labour intensity, and this leads to decreased purchasing power in other fields of the economy.

There are numerous mechanisms through which these indirect effects can operate. For instance, waste management companies may charge higher prices, or municipalities may increase taxes (or reduce expenditures in other areas). If higher public expenditure is met by borrowing, interest rates will increase. If the economy is close to full employment (which is evidently not the case today, but we are analysing here a projection until 2020), a higher labour demand will lead to higher wages in all sectors, which will translate in higher prices, etc…

In other words, the employment effects described above will, to some extent, be “crowded out”. This is precisely why, in a cost benefit analysis, increased employment is a cost to society, not a benefit: it takes away scarce resources (labour) from other valuable applications.

The only assumption under which the scenarios described above could lead to *net* job creation is if the people employed in waste management would not be competitive on the regular labour market. This would for instance be the case if these employees have been unemployed for a very long time and have lost the necessary skills and attitudes. An additional benefit of waste management activities is then that they could lead to the social re-integration of hard to employ people. The question remains open how cost-effective this approach is compared to other labour market policies.

It should be noted that the RPA study referred to above has demonstrated that waste management measures are likely to have only a small effect, either positive or negative, on employment. RPA concluded that the most significant effects may arise outside the directly-regulated sector, making the use of approaches that take account of indirect effects particularly important.

7.13 Lessons Learned

The low financial cost of IVC, combined with its relatively low external environmental cost, make it the best performing option for 16 out of 27 Member States under the private metric (where environmental costs are also considered). The relative environmental benefits associated with the AD options on their own continue to be insufficient to outweigh the influence of the financial cost in the majority of cases under this metric, although the renewable energy support mechanisms do tip the balance in favour of AD for several countries. However, since the support mechanisms are conceptually intending to internalise the environmental costs, it is arguable that this metric may be double counting the environmental impacts.

Under the social cost metric, AD performs better than IVC for 9 of the 27 countries as a result of the environmental benefits and lower discount rate on the cost of capital, notwithstanding the loss in renewable energy support mechanisms.

For both metrics, the current source of electricity is an important factor in determining the net benefits of different switches. The biogas to vehicle option typically performs the best of the different AD options in those countries that generate their energy from the cleaner sources of electricity. The production of vehicle fuel is likely to displace diesel for the foreseeable future. Electricity, on the other hand, is produced from a much more diverse set of resources. Given the increasing emphasis on renewable electricity generation across Europe, it may be expected that a reducing marginal benefit from displacing electricity might lead to increased emphasis on use of biogas for vehicle fuel.

Under the private metric the important variables that influence the level of net benefits of switches to organic treatment options include the level of landfill and incineration taxes, and the renewable energy support mechanisms. Given the more demanding goals for the generation of renewable energy that are likely to be forthcoming in Member States as a result of the Renewable Energy Directive, this is an area where many changes may be expected. To the extent that these would bring about an increase in financial support for renewable energy, this would result in increased support for AD options that would not be available for IVC treatment methods. At the same time, some justification for these support measures ought to be offered, and the analysis suggests that unless, for example, the damages assumed to be associated with CO_2 emissions increase significantly, then only measures which internalise externalities at levels above the central levels used in this study will shift the balance from IVC to AD (by making the environmental benefits large enough to justify the higher financial costs).

8 First policy scenario: only compost standards

In this scenario, it is envisaged that compost standards would be introduced across the EU with a view to giving compost the status of a product. No specific standard has been proposed, though there is a precedent of sorts which arises from the discussions which previously took place in the context of proposals for a Biowaste Directive.[93] In addition, a number of studies have been carried out regarding compost standards in the past, one of these being work undertaken by a consortium on behalf of the European Commission's Joint Research Centre.[94] However, it would appear that only in the consideration of a Draft Biowaste Directive has anyone set out to establish what standard should apply, though one other study has proposed a range of standards based upon a precautionary approach to applications of compost in agriculture.[95]

Many countries already operate their own system of compost standards. Key issues which a study such as this faces, therefore, are:

- What the standard should be?;
- Which countries would be affected by that standard (which would have to change)?
- What would the change imply for those countries?

8.1 Assumptions Regarding the Standard

We have assumed that the standard which would apply is akin to the standard set in the context of discussions for a Biowaste Directive. The standard is effectively two standards, one set at a relatively tight level, the other, at a less stringent level, with certain restrictions applied to the latter. In the following section, we assess the possible implications of the standards for different countries' existing standards if they were to be implemented.

8.1.1 Waste from Separate Collections

In the first instance, the draft Biowaste Directive sought to establish separate collection systems. It is common for compost standards in different countries to specify materials which are either explicitly excluded from the scheme, or explicitly included within the scheme.

[93] EU Commission, DG Environment. Working Document (WD), 2nd draft: "Biological treatment of biowaste"

[94] J. Barth, F. Amlinger, E. Favoino, S. Siebert, B. Kehres, R. Gottschall, M. Bieker, A, Löbig and W. Bidlingmaier (2008). *Compost Production and Use in the EU.* Report for the European Commission DG/JRC.

[95] See F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

The following countries already exclude mixed waste compost from their schemes:

- Austria (for quality compost);
- Belgium;
- Czech Republic;
- Germany;
- Denmark;
- Finland;
- Hungary;
- Luxembourg;
- Netherlands;
- Sweden;
- United Kingdom.

Consequently, these countries would be unaffected by a standard which excluded 'mixed waste' from the scope of standards.

Of those countries which have standards, but do not currently exclude mixed waste from the scope of their compost standards (see Table 8-1), only Spain and France currently produce significant quantities of biowaste treatment products.

In addition, as far as we are aware, the following countries have no standards in place:

- Bulgaria – in the baseline, Bulgaria is assumed to develop biowaste collections with wit waste being composted
- Cyprus – in the baseline, Cyprus is assumed to develop biowaste collections with wit waste being composted
- Greece – in the baseline, Greece is assumed to rely heavily on MBT / mixed waste composting to deliver its landfill directive obligations
- Estonia – in the baseline, Estonia
- Latvia – in the baseline, Latvia is assumed to develop separate collections of biowaste for composting
- Malta -
- Portugal; and
- Slovakia.

If the baseline assumptions are correct, then these countries are in the process of developing industries based around separate collection of biowaste. In this context, experience of other countries suggests that the development of outlets for the products of biowaste treatment will be made easier through a combination of standards, supported by voluntary quality assurance schemes (now operated in many Member States). Indeed, one of the aims of standards is to give potential end-users greater confidence that the material they produce, derived from waste materials, has a status akin to a product. At the same time, because the standard effectively confers a status of a 'quasi-product; nature on the output, it becomes important to ensure that the standard is set in such a way that the materials that have such status conferred upon them are deserving of it.

If standards are set at too relaxed a level, for example, in terms of physical impurities, they may compromise the development of what may be fledgling compost industries.

Table 8-1: General Overview of How Member States Establish Specific Requirements for Input Materials in Composting

Country	**Input Material**							
	Source-Separated						**Municipal Sewage-sludge**	**Not source-separated municipal solid waste**
	Not ABP		**Category 2**		**Category 3**			
	Green waste	**Industry/ commerce vegetable-origin organic**	**Manure**	**Paunch waste**	**Biowaste**	**Industry/ commerce organic residues including. Cat. 3 ABP**		
AT	●	●	●	●	●	●	○	○
BE *Flanders*	●	●	●	●	●[96]	●		
BG	No regulation or standard							
CH	●	●	●	●	●	●		
CY	No regulation or standard							
CZ	●	●	●		●	●	○	
DE	●	●	●	●	●	●		
DK	●	●	●	●	●	●	●	
EE	No regulation or standard							
ES	●	●	●	●	●	●	●	●
FI	●	●	●		●[97]	●	●[98]	
FR	●	●	●	●	●	●	○	●
GR	Currently only mixed waste composting							
HU	●	●	●	●	●	●	●[99]	
IE	●	●	●	●	●	●[100]	●	○
IT	●	●	●	●	●	●	○	○
LT	●	●	●	●	●	●	●	●
LU	●	●			●	●	●[101]	
LV	No regulation or standard							

[96] In Belgium, VFG waste (vegetable, fruit, garden waste) has been collected from households since the early 90's – this has historically excluded any ABP, though two ABPR licensed compost plants now allow general 'food waste' (including ABP) as input materials.
[97] The definition of biowaste in Finland includes plant and animal residues and partly even paper.
[98] Mixed MSW and paunch waste can only be used as a feedstock when co-composted with sewage sludge.
[99] Sewage sludge is allowed, but compost that uses this input material rarely makes the required 'quality grade' for application to agricultural land.
[100] Excludes slaughterhouse wastes
[101] Sewage sludge may only be used when combined with green waste. The permissible input materials are defined within individual plant licences thus can differ from plant to plant.

Country	Input Material							
	Source-Separated						Municipal Sewage-sludge	Not source-separated municipal solid waste
	Not ABP		Category 2		Category 3			
	Green waste	Industry/ commerce vegetable -origin organic	Manure	Paunch waste	Biowaste	Industry/ commerce organic residues including. Cat. 3 ABP		
MT	No regulation or standard							
NL	●	●	●	●	●	●		
PL	●	●	●	●	●	●	●	●
PT	No regulation or standard							
SE[102]	●	●	●	●	●	●		
SI	●	●	●	●	●	●	●	●
SK	No regulation or standard							
UK[102]	●	●	●	●	●	●		

Key: ● *Input material allowed for compost/anaerobic digestion (though where the input material includes ABP, individual plant approval is often required from the veterinary service).*

○ *Input material allowed, but with restrictions for marketing or use, and/or requiring that certain quality requirements are met prior to use e.g. in Italy, sewage sludge limited to maximum 35 % of the starting input mix.*

Sources: Compiled using information from questionnaires, Barth et al. (2008)[1] and the ISWA (2006)[2].

8.1.2 Process Validation Tests

As regards process validation, the draft Biowaste Directive suggested:

An indicator organism shall be used in order to determine the effectiveness of the treatment in sanitising biowaste. This test shall be carried out for each treatment plant within 12 month of its starting up phase.

The test shall be repeated if the composition of the biowaste significantly changes or if major modifications to the process treatment are made.

The indicator organisms shall be Salmonella Seftenberg W775 (H_2S negative) [under review].

These requirements have effectively been superseded by the standards for compost under the EU Animal By-products Regulations.

[102] Note that the input standards are voluntary rather than statutory in this instance.

[1] Barth, J., Amlinger, F., Favoino, E., Siebert, S., Kehres, B., Gottschall, R., Bieker, M., Löbig, A. and Bidlingmaier, W. (2008). *Compost Production and Use in the EU*. Report for the European Commission DG/JRC.

[2] International Solid Waste Association. (2006). *Biological Waste Treatment Survey*. Edited by W. Rogalski and C. F. Schleiss.

8.1.3 Temperature / Time Regimes

As regards hygienisation, the Draft Biowaste Directive suggested the following requirements

	Temp (deg C)	*Treatment Time*	*Turnings*
Windrow	*> or = to 55*	*2 wk*	*5*
Windrow	*> or = to 65*	*1 wk*	*2*
IVC	*> or = to 60*	*1 wk*	*N/A*

As far as we are aware, there is only one instance where the required temperature under a compost standard is slightly less than that specified above, and this is the Czech Republic (45 deg C for 5 d so long as compost does not contain sewage sludge). Also, in Italy and France, the requirement is for 55 deg C for 3 days rather than for 14 days.

However, the ABPR is likely to have had an influence in most countries, both in respect of temperature and time requirements where it is animal by-products that are being composted. It is important to recognise that compost standards do not relate only to animal by-products, but to a range of materials, of which eligible animal by-products may be a sub-set.

For anaerobic digestion, the Draft Biowaste Directive suggested the following:

8. (AD process) The anaerobic digestion process shall be carried out in such a way that a minimum temperature of 55 °C is maintained over a period of 24 hours without interruption and that the hydraulic dwell time in the reactor is at least 20 days.

In case of lower operating temperature or shorter period of exposure:

– the biowaste shall be pre-treated at 70 °C for 1 hour, or

– the digestate shall be post-treated at 70 °C for 1 hour, or

– the digestate shall be composted.

To our knowledge, the temperature and time requirements are slightly lower in Switzerland – 53 deg C for 24 h. In addition, Sweden has a lower dwell time (10 h), but 55 degC must be maintained for 7 d.

8.1.4 Sanitisation Requirements

The sanitisation requirements were set out as follows:

8. (End-product) Compost/digestate is deemed to be sanitised if it complies with the following:

– Salmonella Seftenberg absent in 50 g of compost/digestate [under review]

– Clostridium perfringens absent in 1 g of compost/digestate [under review]

· Compost/digestate shall have less than three germinating weed seeds per litre.

For some of these requirements, the sanitisation requirements under existing compost standards are typically much tighter than this, Salmonella having to be absent altogether in many countries. Rather few countries use Clostridium to demonstrate sanitisation of the output.

8.1.5 Potentially Toxic Elements

The requirements in terms of potentially toxic elements were split into Class 1 and Class 2 (as well as for stabilised biowaste). They are set out in Table 8-2 below.

Prevailing standards for Member States where these are in place are shown in Table 8-3 below. The following observations can be made

- As regards Class 1, most countries would need to tighten their heavy metal standards for cadmium, mercury, lead and zinc to comply with the levels above, including Austria (class A – agricultural use), Germany, Belgium, Czech Republic and Sweden. The requirement is similar to that of organic farming limits really i.e. EC REg. 2092/91.

Table 8-2: Limit Values for Metals and Impurities in the Second Draft Biowaste Directive

Parameter	Compost/digestate (*)		Stabilised biowaste (*)
	Class 1	Class 2	
Cd (mg/kg dm)	0.7	1.5	5
Cr (mg/kg dm)	100	150	600
Cu (mg/kg dm)	100	150	600
Hg (mg/kg dm)	0.5	1	5
Ni (mg/kg dm)	50	75	150
Pb (mg/kg dm)	100	150	500
Zn (mg/kg dm)	200	400	1 500
PCBs (mg/kg dm) (**)	-	-	0.4
PAHs (mg/kg dm) (**)	-	-	3
Impurities >2 mm	<0.5%	<0.5%	<3%
Gravel and stones > 5 mm	<5%	<5%	-

(*): Normalised to an organic matter content of 30%.
(**): Threshold values for these organic pollutants to be set in consistence with the Sewage Sludge Directive.

- Class 2: the following countries would still need to tighten their standards for heavy metals as specified:
    - AT – Class A for agric. (Zinc)
    - CZ – agricultural classification (Cadmium)
    - DK (Copper, Zinc)
    - GR (all)
    - HU (Cadmium)
    - IT (Zinc)
    - SE (Zinc)

- PL (Cadmium, Copper, Zinc

It can be appreciated from the above that the element on which most of the standards would fail is Zinc.

It should be noted that this assessment is based upon a somewhat cursory review of the standards in place in other countries, and that there are usually tolerance bands applied around specific standards. Sometimes, the specification of tighter limit values is accompanied by tolerances which are more generous (see below). As such, without much closer inspection, it could not be demonstrated that the standards implied a much more lax interpretation of what was set out in the Standards above.

Table 8-3: Heavy Metal Limits (mg/kg d.m.) in European Compost Standards

Country	Regulation	Type of standard	Cd	Cr_{tot}	CrVI	Cu	Hg	Ni	Pb	Zn	As
			mg/kg d.m.								
AT	Compost Ord.: Class A+ (organic farming)	statutory ordinance	0.7	70	-	70	0.4	25	45	200	-
	Compost Ord.: Class A (agriculture; hobby gardening)		1	70	-	150	0.7	60	120	500	-
	Compost Ord.: Class B (non agricultural) limit value (only for landscaping; reclamation) (guide value)*		3	250	-	500 (400)	3	100	200	1,800 (1,200)	-
BE	Royal Decree, 07.01.1998	statutory decree	1.5	70	-	90	1	20	120	300	-
BG	No regulation	-	-	-	-	-	-	-	-	-	-
CH	Ordinance on Environmentally Hazardous Substances – limit values for compost and digestate from biowaste	statutory	1	100	-	100	1	30	120	400	-
CY	No regulation	-	-	-	-	-	-	-	-	-	-
CZ	Use for agricultural land (Group one)	statutory									
		Composts without sewage sludge	2	100	-	100	1	50	100	300	10
		Compost with sewage sludge	2	100	-	100	1	50	100	500	10
		Digestates	2	100	-	100	1	50	100	400	10
	Landscaping, reclamation (draft Biowaste ordinance) (group two)	statutory									
		Class 1	2	100	-	170	1	65	200	500	10
		Class 2	3	250	-	400	1.5	100	300	1200	20
		Class 3	4	300	-	500	2	120	400	1500	30
DE	Quality assurance RAL GZ - compost / digestate products	voluntary QAS	1.5	100	-	100	1	50	150	400	-
	Bio waste ordinance	statutory decree									
	(Maximum application 30 t d.m./ha over 3 year period)	Class I	1	70	-	70	0.7	35	100	300	-
	(Maximum application 20 t d.m./ha over 3 year period))	Class II	1.5	100	-	100	1	50	150	400	-
DK	Statutory Order Nr.1650; Compost after 13 Dec. 2006	statutory decree	0.8	-	-	1,000	0.8	30	120/60 for priv. gardens	4,000	25 (priv. gardens only)
EE	Env. Ministry Re. (2002.30.12; m° 87) Sludge regulation	statutory	-	1000	-	1000	16	300	750	2500	-
ES	Real decree 824/2005 on fertilisers Class A	statutory	0.7	70	0	70	0.4	25	45	200	-
	Class B		2	250	0	300	1.5	90	150	500	-
	Class C		3	300	0	400	2.5	100	200	1000	-

Country	Regulation	Type of standard	Cd	Cr_{tot}	CrVI	Cu	Hg	Ni	Pb	Zn	As
			mg/kg d.m.								
FI	Fertiliser Regulation (12/07)	statutory decree	1.5	300	-	600	1	100	150	1,500	25
FR	NFU 44 051	standard	3	120		300	2	60	180	600	
GR	KYA 114218, Hellenic Government Gazette, 1016/B/17- 11-97 [Specifications framework and general programmes for solid waste management]	statutory decree	10	510	10	500	5	200	500	2,000	15
HU	Statutory rule 36/2006 (V.18)	Statutory – also includes Co: 50; Se: 5	2	100	-	100	1	50	100	--	10
IE	Licensing of treatment plants (EPA) *(n.b. no sample shall exceed 1.2 times the quality limit values set)*										
	(Compost – Class I)	statutory	0.7	100	-	100	0.5	50	100	200	-
	(Compost – Class II)	statutory	1.5	150	-	150	1	75	150	400	-
IT	Law on fertilisers (L 748/84; and: 03/98 and 217/06) for BWC/GC/SSC	statutory decree	1.5	-	0.5	230	1.5	100	140	500	-
LT	Regulation on sewage sludge Categ. I (LAND 20/2005)	statutory	1.5	140		75	1	50	140	300	-
LU	Licensing for plants		1.5	100	-	100	1	50	150	400	-
LV	Regulation on licensing of waste treatment plants (n° 413/23.5.2006) – no specific compost regulation	statutory =threshold between waste/product	3			600	2	100	150	1,500	50

Country	Regulation	Type of standard	Cd	Cr_{tot}	CrVI	Cu	Hg	Ni	Pb	Zn	As
			mg/kg d.m.								
NL	*BOOM Compost*	terminated 31/12/2007	*1*	*50*	-	*60*	*0.3*	*20*	*100*	*200*	*15*
	BOOM very clean Compost		*0.7*	*50*	-	*25*	*0.2*	*10*	*65*	*75*	*5*
	Amended National Fertiliser Act from 2008	statutory	1	50		90	0.3	20	100	290	15
PL	Organic fertilisers (includes compost placed on the market)	statutory	5	250		400	3	50	250	1500	15
PT	Standard for compost is in preparation	-	-	-	-	-	-	-	-	-	-
SE	Guideline values of QAS plus sewage regulations for Cu, Zn.	voluntary	1	100	-	600	1	50	100	800	
SK	Industrial Standard STN 46 5735 Cl. 1	voluntary (Mo: 5)	2	100		100	1	50	100	300	10
	Cl. 2	voluntary(Mo: 20)	4	300		400	1.5	70	300	600	20
UK	Standard: PAS 100	voluntary	1.5	100	-	200	1	50	200	400	-
EU ECO Label	COM Decision (EC) n° 64/2007 eco-label to growing media COM Decision (EC) n° 799/2006 eco-label to soil improvers	voluntary [Mo: 2; As: 10; Se: 1.5; F: 200 [only if materials of industrial processes are included]	1	100	-	100	1	50	100	300	10
EC Reg. n° 2092/91	Organic farming requirements	statutory	0.7	70	-	70	0.4	25	45	200	-

*guide/ limit value for Cu and Zn; if the guide value in the compost is exceeded then the concentration has to be indicated on the label.

Source: In D. Hogg, D. Lister, J. Barth, E. Favoino and F. Amlinger Questionnaires and Barth, J., Amlinger, F., Favoino, E., Siebert, S., Kehres, B., Gottschall, R., Bieker, M., Löbig, A. and Bidlingmaier, W. (2008). Compost Production and Use in the EU. Report for the European Commission DG/JRC.

The tolerances and sampling frequencies set in the Draft Biowaste Directive are set out below. It will be appreciated that a 20% deviation for some samples from a standard for Zinc of 400 mg/kg dm would be broadly equivalent to a standard of 500 mg/kg dm where all samples have to be below the required level. As such, a simple reading of the figures does not necessarily tell the whole story.

Table 8-4: Sampling Frequencies and Tolerances in the Draft Biowaste Directive

Series of samples taken in any twelve-month period	Maximum permitted number of samples which fail to conform to any given parameter	Allowed deviation from statutory limit of samples which fail to conform to any given parameter
2	1	20%
4	1	20%
12	3	20%

8.1.6 Quality Assurance Systems

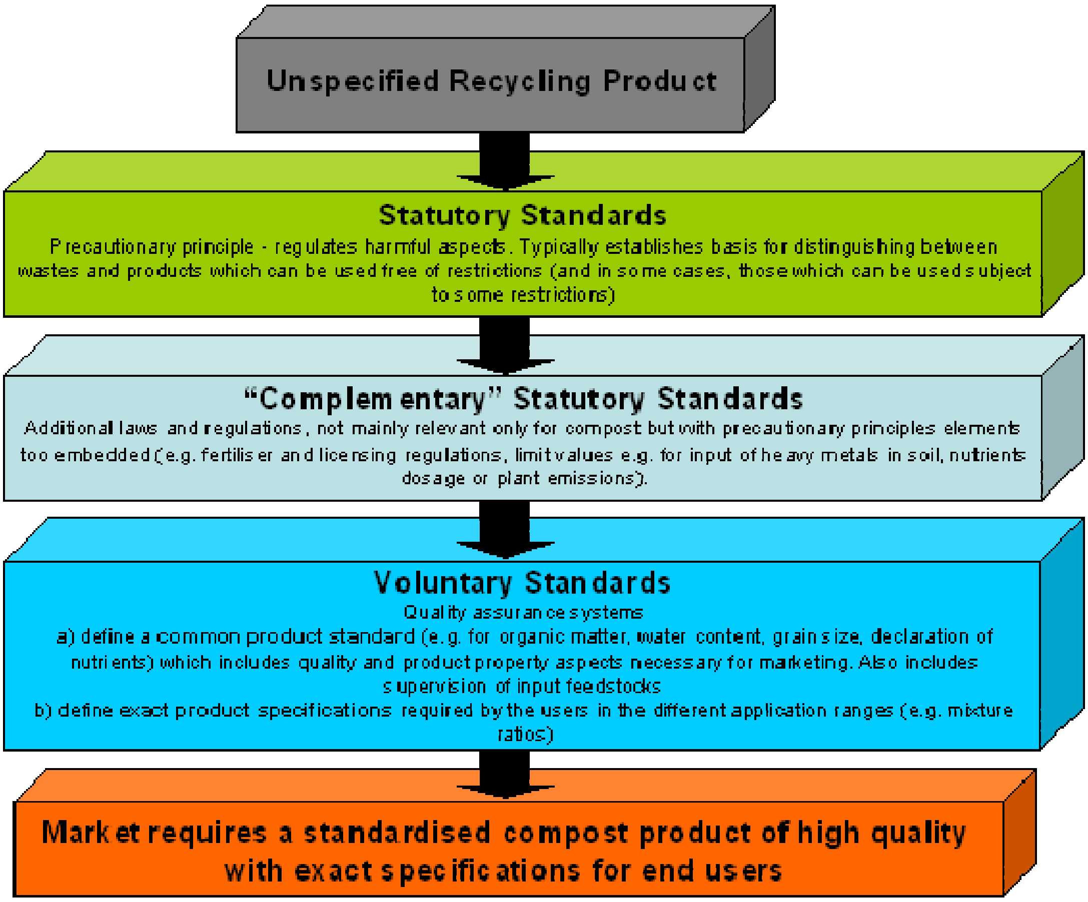
The Draft Biowaste Directive proposed that:

Producers of more than 10,000 tonnes per year of compost or digestate shall implement a quality assurance system for the treatment process."

There are good reasons to believe that the Draft Biowaste Directive was, in this respect at least, somewhat lenient in terms of how it saw the role of quality assurance schemes. Quality assurance schemes have effectively become bodies which deliver on standards and ensure they are implemented. It could be argued that allowing exceptions for relatively large facilities, as the Draft biowaste Directive did, would constitute an incentive for some operators to seek to fall below this threshold. They might thereby gain a competitive advantage in the market place, avoiding the costs of implementing the requirements of a quality assurance scheme, but potentially deriving much of the benefit.

However, the Draft Biowaste Directive showed an appreciation of the inter-relation between the two (standards and quality assurance). This relationship is highlighted below in Figure 8-1.

Figure 8-1: Range of Compost Standardisation – from Output Material to Marketed Product



Source: D. Hogg, J. Barth, E. Favoino, M. Centemero, V. Caimi, F. Amlinger, W. Devliegher, W. Brinton, and S. Antler (2002). *Comparison of Compost Standards Within the EU, North America and Australasia*. Report for the Waste and Resources Action Programme.

It might be appreciated, from the above discussion of compost standards, that such standards are generally oriented to 'preventing the negative' consequences of production and use of compost. Standards can *underpin* market development, but they might not, on their own, help to increase demand. Indeed, some commentators argue that the focus on such standards can result in a focus on 'negative aspects' of composts rather than on their positive attributes.

There are a number of means through which demand for use of compost can be increased. In the important agricultural market, however, farmers need to be able to trust the materials they are using. Where composting is a fledgling industry, farmers may have little or no experience of dealing with composts, especially those including food waste in the feedstock, and in the context of a market in which a range of products are available, some guarantee of quality and performance is important. Quality assurance systems focus more on the positive attributes of compost, how to give comfort to end-users around the use of the material, and how awareness of these can be promoted in the market place.

Quality assurance systems (QASs) have played a key role in improving the status of compost in the eyes of end users in a number of the different EU Member States. QASs seek to make the link between compost / digestate production and the markets for compost / digestate application. As outlined in a report for WRAP,[103] QASs ‘start’ where statutory standards following the precautionary principle normally end. Statutory standards have tended to focus on the prevention of harm, or the avoidance of negative consequences from the application of compost / digestate. It is rare for statutory standards to address themselves to issues related to specific markets for compost / digestate application. Exceptions here are sewage sludges and products used in organic agriculture. For sewage sludges, the regulations concerning applications are frequently prescribed (as, to some extent, in the UK through the Sludge Use in Agriculture Regulations). For products used in organic agriculture, the product standard in terms of concentrations of potentially toxic elements is closely related to the end-use application. For other products though, QASs close the recycling loop for (usually source-separated) organic residues.

QASs focus on meeting the demands of end-users. In this sense, where statutory standards are in place, QASs complement these. In countries where no, or only very limited statutory standards exist, QASs are important in the recovery of organic waste because they can seek to control quality at all stages of the treatment of organic residues, such as:

- **Separate collection / quality of feedstock**
  Quality assurance may require that inspections on the quality of feedstock are carried out frequently to ensure that end products are of the desired quality, and have the desired characteristics. Such inspections can be used to suggest measures for improvement (for example, in approaches to source-separation). At a more practical level, they might also support compost / digestate producers as they seek to ensure that batches received for treatment are suitable for the process (if they are not, loads may be rejected). Finally, they may assist in traceability of batches;
- **Plant engineering**
  Errors in the plant engineering can be quickly identified via quality controls. Regarding the issue of hygiene, quality assurance also serves to guarantee worker protection;
- **Compost production**
  Only regular or continuous process monitoring and recording as well as constant quality and product checks can ensure errors in compost production are avoided;
- **Marketing**
  End-users, including farmers, are likely to seek a standardised quality compost / digestate. Typically, this is guaranteed by the quality assurance system. Statutory standards are useful here since they require testing and evaluation before it can be determined whether the material is of acceptable quality. A QAS improves confidence levels that the product offered is *consistently* of a specified quality *and* conforms to statutory requirements. An associated quality symbol can lend support to any marketing efforts of the compost / digestate producer;

[103] D. Hogg, J. Barth, E. Favoino, M. Centemero, V. Caimi, F. Amlinger, W. Devliegher, W. Brinton, and S. Antler (2002). *Comparison of Compost Standards Within the EU, North America and Australasia*. Report for the Waste and Resources Action Programme.

- **Public relations work**
  In order to improve the public perception of compost / digestate, some public relations activity is important. For each individual compost / digestate producer to undertake this would be expensive. QASs benefit, in this respect, from economies of scale to the extent that as their membership grows, a more concerted PR campaign can be developed at reduced costs to *individual* producers. A positive image for compost / digestate can be developed on the basis of assured quality, and through use of a quality symbol for the compost product;
- **Application**
  Under QASs, it is typical to require the characteristics of a given compost / digestate to be declared so that end-users can understand the appropriateness, or otherwise, of a given compost / digestate for their purposes. Hence, analytical tests are carried out, the results of which form the basis for this declaration and associated recommendations for use. This is crucial in order for the end-use market to make the correct decision about the quality nature of the compost / digestate, its appropriateness for the application in question, and the appropriate application rate in the context of that application;
- **Product range**
  In the ideal case, QASs develop a range of products with specific characteristics, more-or-less tailored to specific end-use markets. QASs can do this to the extent that they understand, as a result of analytical testing, the properties of compost / digestate and the extent of their fluctuation in well-operated plants. This 'variance' can also help end-users make their decisions about appropriate products for use;
- **Policy/regulation**
  Through statistical evaluation of the test results, the legislators are made familiar with the present standard of compost / digestate and the 'performance frontier' of composting / digestion plants. Such data can be used to inform the development of policies and regulations that are appropriate for the current practical situation. Indeed, as discussed in a WRAP report, many countries' pragmatic approach to setting standards has been informed by a desire to apply the precautionary principle, but in a manner contextualised by 'what is actually possible'.[104] As such, this type of data is extraordinarily useful in understanding the key characteristics and possibilities of composting / digestion plants, as well as the characteristics of composts / digestates and their variation with changes in the input feedstocks;
- **Certification**
  A quality assurance system is a pre-condition for the certification of composting plants, e.g. the EU-Standard ISO 9000 and ISO 14000.

Besides these points, all marketing analysis over recent years shows that users of compost demand a standardised quality product that is verified by independent organisations. A study in the south of Germany showed that *94% of the commercial users* were making this a pre-condition of use.

Market research carried out in the state of Lower Saxony in Germany concerning the expectations of the green sector regarding compost led to the apparently contradictory result that the quality symbol seems to be relatively unimportant in their eyes (see Table 8-5). However, on reflection, it can be appreciated that whilst other elements were rated

[104] D. Hogg, D. Lister, J. Barth, E. Favoino and F. Amlinger (2009) *Frameworks for Use of Compost in Agriculture in Europe*, Final Report for WRAP, January 2009 http://www.wrap.org.uk/downloads/Eunomia_compost_in_agriculture_final_report.703534d2.6993.pdf .

as being of higher priority than a QAS symbol, these elements are, in fact, themselves, usually an integral part of any QAS (or are implied by the existing regulatory standards). By encompassing all such parameters, QASs ought to give comfort to end-users concerning these issues. Indeed, an upshot of the study was a new communication strategy in the German Compost Quality Assurance Organisation BGK which clearly pointed out what the quality assurance system contains and what stands behind the symbol.

Table 8-5: Expectations of the Green Sector for Compost Products

Percentage of interviewed persons	**Requirement**
65%	Compost use should not create health problems
64%	Low content of heavy metals
61%	Analysis by approved labs
56%	No impurities (glass, stones...)
52%	No seeds in the compost
48%	Information about raw material
43%	Good declaration of nutrients
40%	Recommendations how to use
36%	**Compost has a quality symbol**
35%	Origin of the compost

Source: Hogg, D., Barth, J., Favoino, E., Centemero, M., Caimi, V., Amlinger, F., Devliegher, W., Brinton, W. and Antler, S. (2002). Comparison of Compost Standards Within the EU, North America and Australasia. Report for the Waste and Resources Action Programme.

The introduction of separate collection and composting should preferably go hand-in-hand with the introduction of statutory standards or, much less preferably, a voluntary quality assurance system. Countries advanced in their experience of composting have recognised this and have developed systems or are preparing them at present. The more advanced QASs tend to be supported by statutory standards. An exception is the Swedish system and the new voluntary certification scheme in the Netherlands.

Participation in full quality assurance schemes is, in all operating countries except Belgium/Flanders, a voluntary act. However, if the quality standard has established itself (and especially if it is statutory), the market begins to *demand* these qualities and composting plants come under pressure to furnish proof of quality (this is very much evident in Germany and the Netherlands).

In the UK, the effect of the Quality Protocol for Compost and the PAS100 (as the only approved standard under the Protocol at the time of writing) is likely to lead to production being increasingly oriented to meet PAS100, and to an increase in demand for compost which meets that standard. The effect of the proposed quality protocol for the Production and Use of Quality Outputs from Anaerobic Digestion of Source-Segregated Biodegradable Waste and the proposed PAS110 standard might be expected to have the same effect in terms of digestate, with the implications being potentially more profound, as the system has been developed so early in the development of the industry for AD of source-segregated biodegradable wastes in the UK.

8.1.7 Summary

In summary, many of the countries which are more advanced in the development of their standards, and in respect of their development of supporting quality assurance systems, would, most likely, not be greatly affected by a standard of the nature proposed. These countries include:

- Austria (for quality compost);
- Belgium;
- Czech Republic;
- Germany;
- Denmark;
- Finland;
- Hungary;
- Luxembourg;
- Netherlands;
- Sweden;
- United Kingdom.

In addition, some countries, such as Ireland, have implemented quality standards akin to those in the Biowaste Directive through their licensing procedures. Ireland is also known to be developing its own standard through its Market Development Programme.[105]

Some countries allowing mixed waste within their standards would be affected. Of those that have infrastructure in place already, France and Spain seem most likely to be affected by a requirement to change their standards to reflect the Draft Biowaste Directive.

Several countries have, as yet, no standards for compost, but the majority are developing their biowaste treatment industries. Some, however, already have some significant infrastructure in place which could be affected by such standards.

Most countries with well-developed markets, or who have the intention to develop such markets, for composts / digestates have QASs in place.

8.2 The Effect of Standards

It is extremely difficult to quantify the benefits of implementing a set of standards for compost. The effects of any policy change have to best against the counterfactual - what would have happened in the absence of standards? In those countries with standards and QASs in place, it would seem that the role has been to increase the level of demand for compost / digestate through increasing appreciation of the potential benefits of their use among key end-users. The effect is particularly relevant in agriculture, which appears still to constitute the major end-use market for compost. A recent report noted the probable effect of standards and QASs in ensuring that few problems have arisen in respect of compost use in agriculture, noting with specific reference to Germany:[106]

[105] Munoo Prasad and Percy Foster (2009) Development of an Industry-led Quality Standard for Source-Separated Biodegradable Material Derived Compost (2006-WRM-DS-26), *STRIVE Report No. 22*, Wexford: EPA.

[106] D. Hogg, D. Lister, J. Barth, E. Favoino and F. Amlinger (2009) *Frameworks for Use of Compost in Agriculture in Europe*, Final Report for WRAP, January 2009 http://www.wrap.org.uk/downloads/Eunomia_compost_in_agriculture_final_report.703534d2.6993.pdf .

Continuing development of trust between compost producers and farmers, success stories within the farming sector, and the strictly quality-oriented work of BGK all helped build confidence within the agricultural sector on the use of compost. Nevertheless, it took nearly 10 years before the first generally available brochure for compost application in agriculture was published. Confidence was, nonetheless, sufficient that even when animal protein was found in sugar beets and the sugar beet organisation subsequently forbade their member farms to use compost (a loss of 30% of the German compost market), the sugar beet industry remained open to discussions with the compost sector, and to the outcomes of research, which finally identified the source of the protein as coming not from compost, but from rodents living on the sugar beet storage piles at the border of the fields.

Only over the last 4 to 5 years has the agricultural sector begun to appreciate fully the positive benefits associated with the use of compost. Supported by discussions at a number of annual conferences in Germany, quality compost is no longer seen to pose a risk to agricultural producers. The challenges that the agricultural sector now faces, such as the need for soil organic matter, humus management and the recovery of nutrients such as phosphorus and nitrogen, all tend to support the need for close cooperation with the compost/digestate sector. In particular, increasing mineral fertiliser prices (up 50% in the last 3 years) during dry summers, where yield differences between crops fertilised with or without compost have been easily detectable on account of the increased water holding capacity, have highlighted the advantages of compost for farmers on a practical level.

The positive perception is, for example, demonstrated by the fact that the sugar beet industry in Germany now requires quality assured compost, and that quality compost is listed in the list of material permissible for use by the German Organic Farmers Organisation.

Another approach to answering this question might be based upon why standards emerged in the first place. For many countries, the first steps along the road of biowaste treatment involved biological treatment of mixed waste. Although the treatment of the waste may have proved possible, deriving a useful product from the mixed waste stream proved more problematic. Compost derived from mixed municipal wastes tended to suffer from problems of physical impurities, and were shown to have higher concentrations of potentially toxic elements than those materials derived from source separated biowaste.

This effectively gave the products of composting a very poor image in the eyes of potential end users. It is important to understand how most markets work in modern societies. Increasingly, transactions take place between people – buyers and sellers - who do not know each other (they will probably never meet) and who, therefore, have to base their exchange on the basis of trust. There are clearly some exceptions to this where those marketing products and service market directly to people and seek to develop relationships based upon trust. However, beyond these exceptions, 'trust' becomes synonymous with being able to use a product without fear of it failing to perform in the desired way. Herein lies the basis of the desirability of product standardization.

Product standardization is not, in and of itself, sufficient to guarantee the success of a product or service in the market place. There is clearly little to be gained by consumers indulging in the practice of purchasing products which are standardized, but only at a low level of quality. Indeed, if other products are operating against the background of a standard which guarantees only a low level of quality, it may be difficult – albeit not impossible – for products to differentiate themselves from other competing products.

Indeed, it may become necessary to distance the product from the standard, or to develop a brand which seeks to establish a unique standard in the marketplace.

Compost, however, is not a high value product which can be the focus of enormous marketing budgets. Rather, it tends to be sold at relatively low values, it is not a material which – because of its relatively low value and considerable bulk – will generally travel long distances to markets (even though, occasionally, the waste delivered to the facility may travel some distance), although where demand for organic matter is high, this may indeed occur. As such, the suppliers of waste-derived composts at the quality end of the market tend to benefit from the recognition that comes from achieving a standard that does, indeed, guarantee that the material meets some fundamental criteria in respect of the material's quality.

Given the above, it seems reasonable to argue that standards, supported by QASs, have a role to play, as long as they are not set at levels which allow for products of low quality to compromise the prospects of those of a higher quality achieving favourable recognition in the market place. In this context, as many new member States seek to develop their own compost markets, the role of standards may be of some significance.

8.2.1 Effects on Different Countries

As noted above, there are many countries where the requirement to implement standards would be expected to have limited effect. In others, where composting / digestion of mixed municipal waste leads to the production of outputs which can be used in agriculture, the effect would be expected to be potentially significant.

It is not clear, however, what the consequence might be in those countries, such as France, where mixed waste composting / digestion is more prevalent. In principle, a number of possibilities present themselves:

1. Because the infrastructure used to develop composts / digestates from mixed wastes is, by and large, similar to that used for composting / digesting source segregated materials, then in principle, the switch from one to the other could occur, as long as separate collection systems were implemented. This switch might not be without some level of redundancy in the treatment technology, but significant redeployment should be possible;
2. Similarly, with limited adaptation of the technology, these facilities could become mechanical biological treatment facilities, focusing, for example, on stabilizing waste prior to its being landfilled, or orienting product development towards low grade applications such as eroded soils and landfill cover; and
3. In more extreme cases, the facility could become redundant if the view is taken that the mixed waste material should simply be incinerated.

Each of these is possible, and the outcome will depend upon local circumstances, the nature of targets which prevail locally, and possibly also, the remaining life of the existing facilities.

In cases where the switch is from mixed waste compost to source segregated materials, then we can gain some information on the likely environmental effects of this through previous work undertaken on behalf of the European Commission.[107]

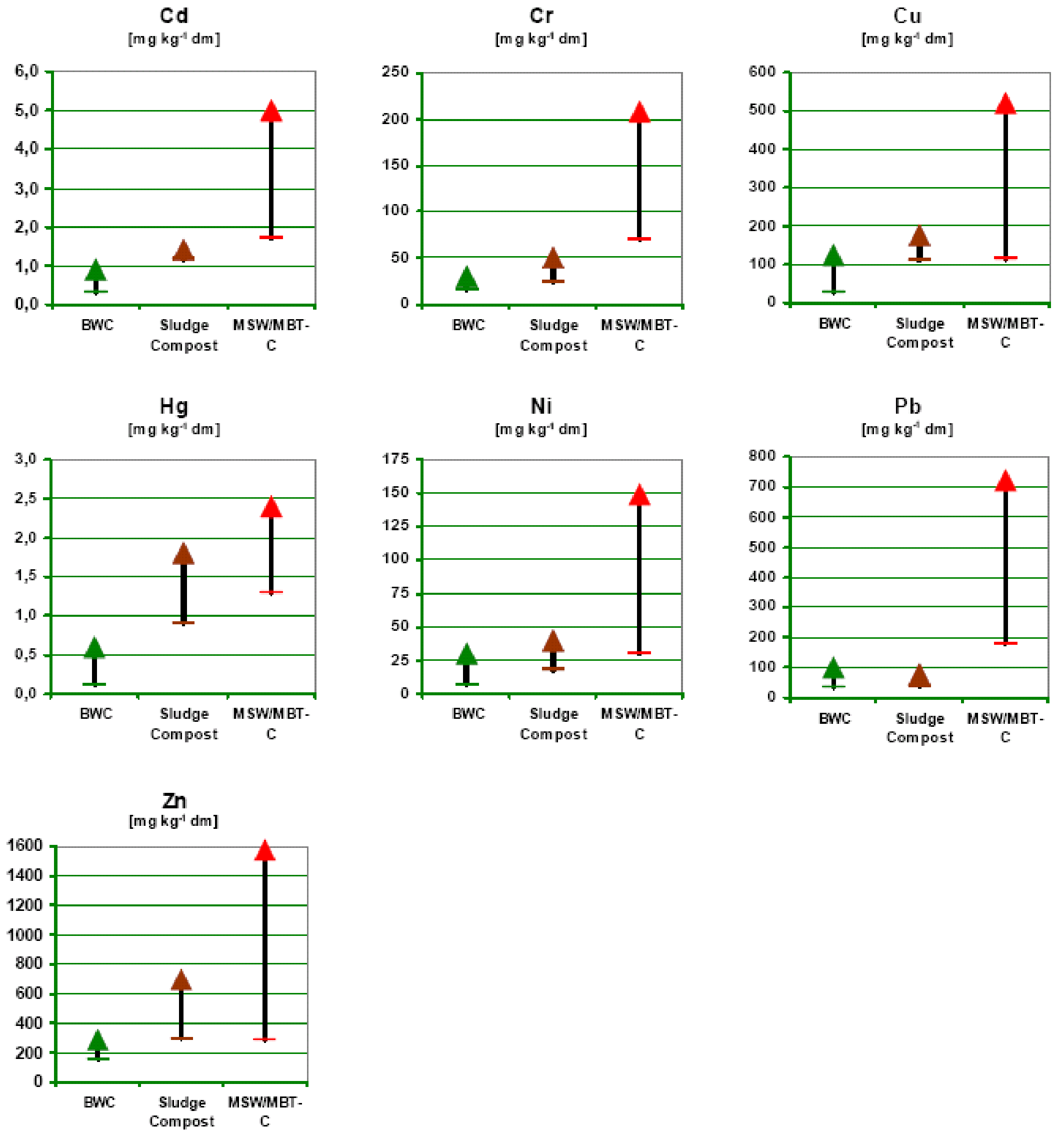
[107] See F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

Figure 8-2 shows the difference in the concentrations of potentially toxic elements found in composts derived from different feedstocks. It clearly demonstrates the fact that for many of the metallic elements, the concentrations are significantly higher for mixed waste composts / residues from MBT plants than they are for source segregated biowastes. This implies, therefore, that to the extent that an absence of standards allows such processes to proceed, they implicitly allow for the spreading on land of material with a higher load of potentially toxic elements. A similar picture is found from another source (see Figure 8-3).

The effect does not appear to be confined to metallic elements. Although several studies note the presence of organic pollutants in compost derived from source segregated wastes, these can usually be traced directly back to factors which have affected the biowaste itself (rather than suggesting some form of *de novo* synthesis in the process).[108]

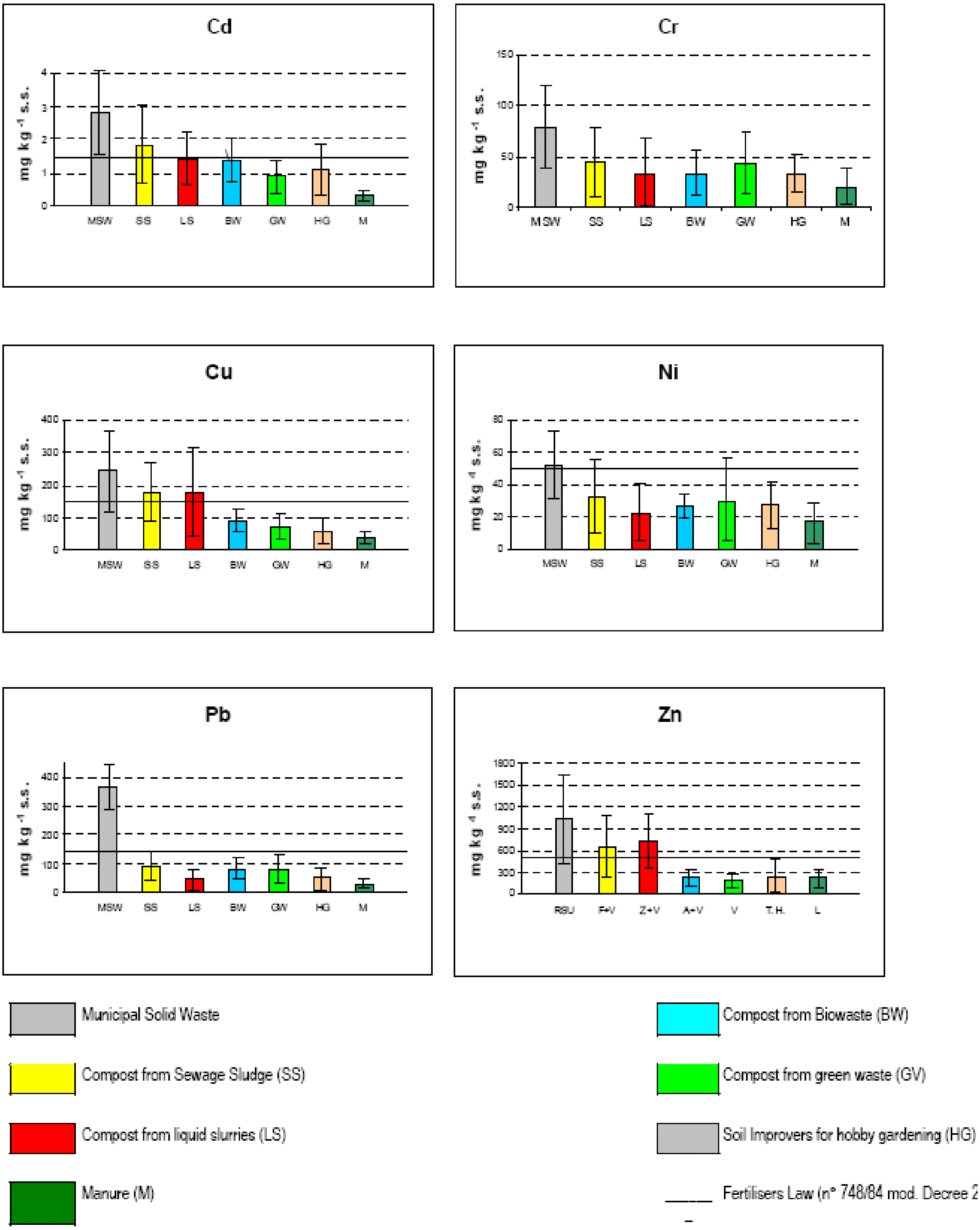
[108] Riedel, H. and Marb, C. (2008). *Heavy Metals and Organic Contaminants in Bavarian Composts – an Overview*. Compost and Digestate: Sustainability, Benefits, Impacts for the Environment and for Plant Production, Proceedings of the International Congress CODIS February 27-29, 2008. Stäb, J., Kuch, B., Rupp, S., Fischer, K., Kranert, M. and Metzger, J. W. (2008). *Determination of Organic Contaminants in Compost and Digestates in Baden-Württemberg, South-West Germany.* Compost and Digestate: Sustainability, Benefits, Impacts for the Environment and for Plant Production, Proceedings of the International Congress CODIS February 27-29, 2008. Timmermann, F., Kluge, R. and Bouldan, R. (2008). *Sustainable Use of Compost in Agriculture - Beneficial Crop Cultivation Effects and Potential Risks*. Research Results of a Long Term Study in Germany, Final Report. Deller, B., Kluge, R., Mokry, M., Bolduan, R. and Trenkle, A. (2008). *Effects of Mid-Term Application of Compost on Agricultural Soils in Field Trials of Practical Importance: Possible Risks*. Compost and Digestate: Sustainability, Benefits, Impacts for the Environment and for Plant Production, Proceedings of the International Congress CODIS February 27-29, 2008. Kuch, B., Rupp, S., Fischer, K., Kranert, M. and Metzger, J. W. (2007) *Determination of Organic Contaminants in Composts and Digestates in the State of Baden-Württemberg, Germany,* Forschungsbericht FZKA-BWPLUS, Förderkennzeichen BWR 240246. Kupper, T., Brändli, R. C., Bucheli, T. D., Stämpfli, C., Zennegg, M., Berger, U., Edder, P., Pohl, M., Niang, F., Iozza, S., Müller, J., Schaffner, C., Schmid, P., Huber, S., Ortelli, D., Becker-Van Slooten, K., Mayer, J., Bachmann, H-J., Stadelmann, F. X. and Tarradellas, J. (2008) *Organic Pollutants in Compost and Digestate: Occurrence, Fate and Impacts*. Compost and Digestate: Sustainability, Benefits, Impacts for the Environment and for Plant Production, Proceedings of the International Congress CODIS February 27-29, 2008.

Figure 8-2: Ranges of mean heavy Metal Concentrations in Biowaste Compost (BWC), Sludge Compost and Compost from Municipal Solid Waste / Mechanical Biological Treatment (MSW / MBT)



Source: F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

Figure 8-3: Heavy Metal levels of Soil Improvers (compost and manure) from different sources compared to Italian Fertiliser Law limit values



From: Centemero M. modified, 2000

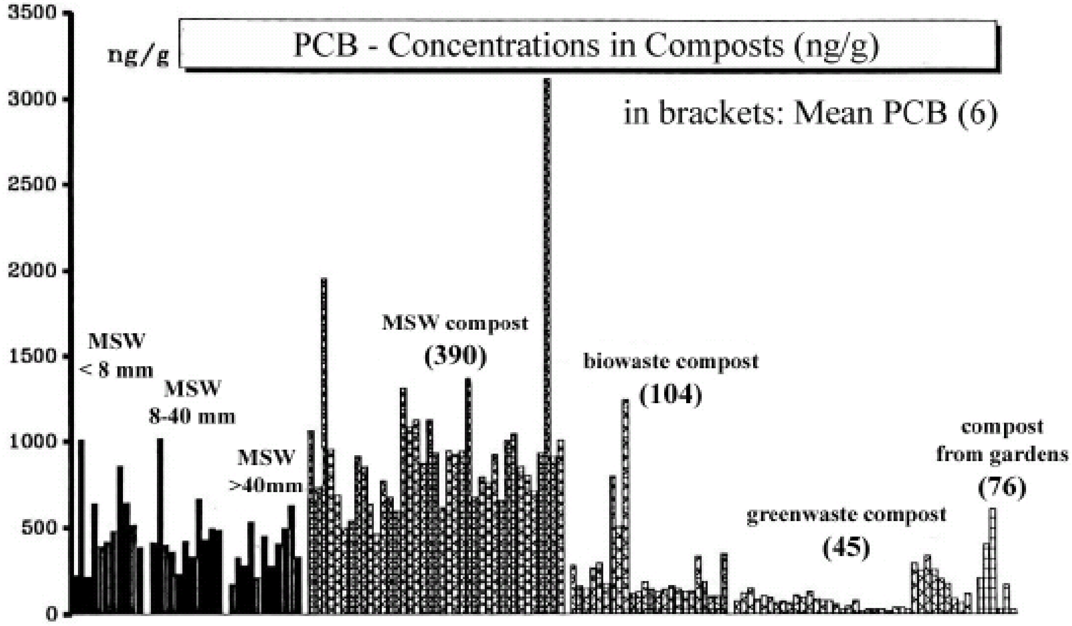
Source: F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

One study noted:[109]

The PAH or PCB concentrations can sometimes be relatively elevated, probably due to inputs from traffic emissions, ashes, or impurities. Specific contamination might also occur due to pesticide application. Urban feedstock and compost was generally elevated in PAH, PCB, and PCDD/F levels in comparison with rural samples. This corresponds to the pattern observed in other environmental compartments. Seasonal differences were observed for PAHs, PCBs, and PCDD/Fs, the concentrations generally being highest during summer. This is in accordance with the seasonal variation observed in the environment for PCBs, but not for PAHs and PCDD/Fs.

The relative concentrations of PCBs and PCDD/PCDF in composts, related to feedstock, are shown in Figure 8-4 and Figure 8-5. Again, this highlights the fact that composts from mixed waste feedstocks show higher levels of organic pollutants than those derived from source segregated materials.

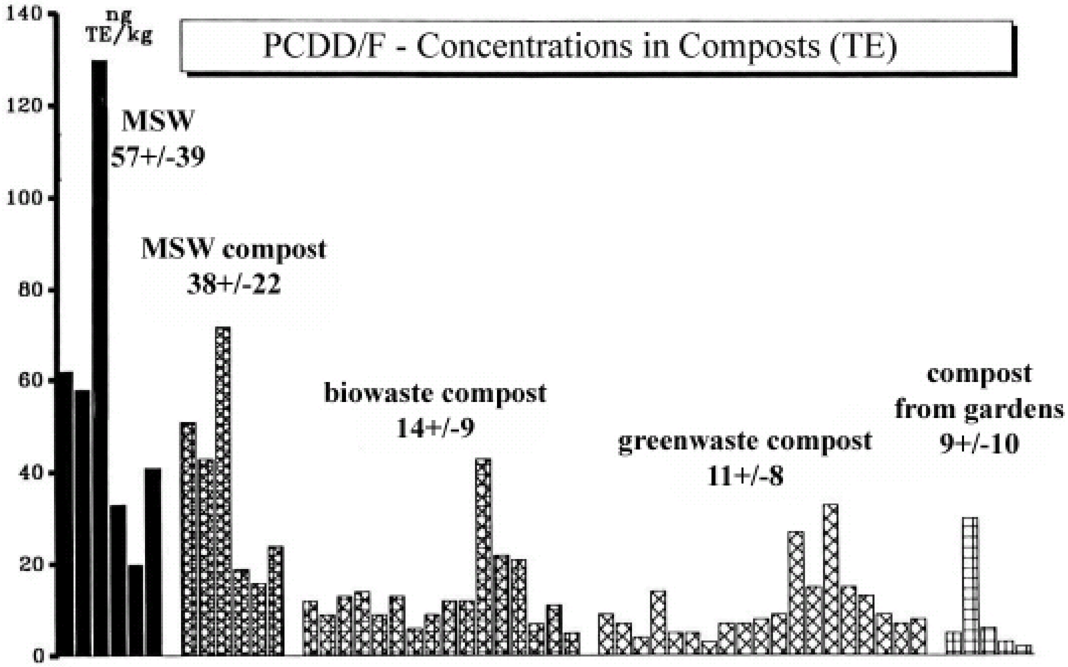
Figure 8-4: Total PCB concentrations in different compost types and mixed solid waste; in addition the average sums of the congeners 28, 52, 101, 138, 153 and 180 (PCB(6)) are shown in brackets (Krauß et al., 1992)



Source: Krauß, P., Krauß, T., Hummler, M., Mayer, J., 1992. Untersuchung von Grün, und Biomüllkomposten auf ihre Gehalte an organischen Umweltchemikalien. Bioabfallkompostierung III, Ministerium für Umwelt, Baden-Württemberg. Boden-Luft-Abfall, Heft 20, 60p, cited in F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

[109] Brändli, R. C., Bucheli, T. D., Kupper, T., Furrer, R., Stadelmann, F. X. and Tarradellas, J. (2005) Persistent Organic Pollutants in Source-separated Compost and its Feedstock Materials – A review of Field Studies, *Journal of Environmental Quality*, Volume 34, May-June 2005, pp735-60.

Figure 8-5: PCDD/F concentrations in different compost types and mixed solid waste; numbers: mean values and standard deviations (Krauß et al., 1992)



Source: Krauß, P., Krauß, T., Hummler, M., Mayer, J., 1992. Untersuchung von Grün, und Biomüllkomposten auf ihre Gehalte an organischen Umweltchemikalien. Bioabfallkompostierung III, Ministerium für Umwelt, Baden-Württemberg. Boden-Luft-Abfall, Heft 20, 60p, cited in F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

In the cases both of metals, and organic pollutants, one would expect that the elevated concentrations of these in composts derived from mixed waste feedstocks would increase the concentrations of these in the soil, thus reducing soil quality relative to the situation which might prevail where composts are derived from source segregated feedstocks. This is most clearly shown in Table 8-6 which shows results from work by Amlinger et al, suggesting that precautionary thresholds for concentrations of specific metals in sandy soils are exceeded far more quickly when using mixed waste as a feedstock (MSW compost) than when using composts derived from source segregated biowastes (Biowaste compost).

Table 8-6: Years until precautionary threshold values for sandy soils are exceeded. Comparison between Biowaste compost and compost derived from biological treatment of residual mixed waste

	Heavy metal contaminants						
	Cd	**Cr**	**Cu**	**Hg**	**Ni**	**Pb**	**Zn**
	Years						
Biowaste composts	281	>500	156	448	>500	319	100
MSW compost	50	57	27	25	30	54	23
Notes: Time span may vary when the assumptions of soil background concentrations are changed							

Source: F. Amlinger, E. Favoino, M. Pollak, S. Peyr, M. Centemero and V. Caimi (2004) *Heavy metals and organic compounds from wastes used as organic fertilisers*, Study on behalf of the European Commission, Directorate-General Environment, ENV .A.2, http://europa.eu.int/comm/environment/waste/compost/index.htm

The longer-term consequences of this are not clear, but it is known that different crops take up different metals from the soil at differing rates, with factors such as moisture, pH and soil type clearly playing a role. Other impacts upon soil and water courses could not be ruled out. Effectively, this renders it more difficult to deliver benefits associated with applying organic matter to soils without at the same time causing potential problems of worsening soil quality.

8.3 Summary

There is a considerable body of evidence which supports the view that composts derived from source segregated materials are likely to have much lower levels of contamination from potentially toxic elements, such as metals and organic pollutants. One of the roles of compost standards is to play a ‘defensive role’ on the part of the environment in seeking to ensure that what is applied to land minimises the likelihood of applications of compost leading to a build up of these elements in the soil such that they reach levels where the land might not be considered fit for cultivation of agricultural crops. It is very difficult to place a monetary value upon the environmental benefits, other than through seeking to understand clean-up costs should this situation emerge. It might be noted, however, that prices paid for quality compost are typically zero (in agriculture) or positive, whereas prices are often negative for materials derived from mixed waste (the producers of these materials pay to ensure an outlet).

In addition, standards, when coupled with QASs, can assist in the development of markets for compost where the standards are set in such a way that those outputs meeting the standard can effectively give reassurance to end-users that the compost / digestate is of a specified quality. This, in conjunction with the development of separate collection of biowaste, has demonstrated its effectiveness in creating the necessary confidence on the part of consumers, a precondition for the acceptance of in the waste derived compost material and composting as a whole.

Standards, therefore, play a dual role:

- They help protect the environment through implementing what is effectively a precautionary approach to the regulation of compost and its application; and

- They can constitute part of the system whereby the producers of compost and digestate can develop a more sound marketing strategy in the face of what are often negative perceptions of compost, and where some potentially important end-users may have little familiarity of the material.

In the context of the Waste Framework Directive, there has clearly been discussion ongoing regarding the possibility of setting a standard for compost which would delineate when compost / digestate would be considered legally ‘no longer a waste’, thereby allowing the outputs which met a high standard to be considered as ‘’a product’. This would, in principle, allow it to be marketed freely across the EU, but would also allow the product to be distinguished from the ‘waste’ from which it was derived, and from lower qualities which have to stay under the waste regime in order to minimise any risks.

It has to be noted that some of the scenarios for some of the Member States include, to varying degrees, the development of source separation and composting across the country. Such a development is difficult to anticipate occurring in the absence of a well developed system of standards, supported by QASs to help develop marketing of the end product. As such, the scenarios do, themselves, appear to imply, or necessitate, the development of some form of coherent marketing strategy within those Member States for whom the swift evolution of source separation is foreseen. Indeed, as regards Scenarios 2 and 3, but especially with regard to Scenario 2, it seems unlikely that these scenarios could materialise without standards and QASs being developed in the countries concerned. Indeed, investments in collection and treatment infrastructure have to be made swiftly in many Member States to meet Landfill Directive targets. Standards allow an informed decision to be made regarding the appropriate balance of separate collection and dedicated biowaste treatment infrastructure, or mixed waste treatment through MBT. This suggests rather that the swift development of a standard might be a pre-condition for the baselines to be achieved.

There is, therefore, something of a chicken and egg situation with respect to standards and QASs, and with respect to the role of separate collection in delivering Landfill Directive targets. Where scenarios for Member States (projected to 2020) include the development of source separation, these policy components included within Scenario 1 (as well as the other scenarios considered) effectively become a pre-requisite (at least nationally) for such a change.

Finally, it is worth highlighting the fact that at the EU level, a ‘weak’ standard might be less valuable than none at all. To the extent that producers of compost and digestate are to be equipped with tools which enable them to market their products with greater confidence, it seems reasonable to argue that they should be able to do this only for materials which achieve a relatively high quality threshold. Marketing products which may be of low quality is likely to be no more effective in markets for compost than it is in markets for cars, not least where so many alternatives are available. On the contrary, allowing products of low quality to be marketed without differentiating them from higher qualities can be confusing for consumers and risks jeopardising the development of a market for compost, and a positive image for it.

Table 8-7:

No EU standard scenario	**EU Standards including quality assurance scenario** **"No need for a standard without control -**
There is no time for a trial and error approach in the MS what the right standard and level is if they want to meet the landfill targets in time.	A European definition what a good quality will help the whole biowaste sector and increase the organics recycling development.
Investments into infrastructure and treatment types/processes of organic waste have to be made now in starting Member States. Standards allow to make the decision where to go and how to get there (separate collection or mixed waste treatment or MBT..)	A high quality and controlled compost standard creates confidence in compost and composting by customers and controlling authorities
All investment in the past in mixed waste composting failed because of a lack of markets in the long run, so composting can not contribute to meet the landfill directive in this case.	Environmental impacts don't stop at borders. Only a European compost standard can reflect the Commission's soil protection strategy.
No standard = no sustainable markets because the compost quality is sold which the treatment plants produce and not what the market accepts. Consumers will test the compost only once. Disappointed customers will create a negative image for compost, for composting and limit their effort in good separate collection and will not come back. So organic waste will end up in costly incineration plants instead of being recycled. The investment in mixed waste plants will be in vain.	Compost costumers act on a European level (Certified food production chains, producers of growing media and soil improvers) and need a uniform and high European compost standard in the long run. Only this allows to explore the full market potential for recycled organics, nationally and internationally.
Waste management thinks in quantities, compost customers in qualities	
None of the countries with low or no compost standards can show a sustainable large compost market until now.	
Low standards kill high quality composting approaches because of the already suspicious image of a waste derived product.	

Summary: The No Standards Scenario" delays the developments for organic waste treatment and diversion from landfills, promotes incineration, increases MSW treatment costs and leads to more environmental damages in the Member States.	**Summary.** A controlled high quality standard establishes organics and compost as successful elements of the intended EU Recycling Society and facilitates composting. A European definition what a good quality is will help the whole biowaste sector and establish compost markets sustainably.

9 Second policy scenario: high prevention and recycling

9.1 Definition and methodology

The second policy scenario is determined by, on the one hand, general assumptions that apply to every policy scenario (see Section 6.2) and, on the other hand, specific assumptions on prevention and recycling targets that will be met in this policy scenario 2.

We consider two variants to this policy scenario, which differ in the assumptions concerning the treatment of separately collected food waste.

9.2 Impact assessment - Scenario 2

The specific assumptions are:

- The waste generation will be reduced compared to the baseline, as a result of effective waste prevention.
- The costs of the preventive actions will not be included quantitatively in the model, as it is not defined which preventive actions will be taken at which cost in which country. We are not looking at the policy itself, but at the policy result, being a reduction in waste generated.
- Home composting will take place in the same proportion as in the baseline policy scenario, it will contribute to reaching the recycling targets.
- The targets for separate collection are 60% of kitchen waste (food waste) and 90% of green waste.

The approach is explained in detail in Annex B to this report, which also contains the country level results.

The following sections identify the changes in costs and benefits, as well as greenhouse gas impacts, of Scenario 2. Table 9-1 summarises the scenario considered within the analysis.

Table 9-1: Key Assumptions Underpinning Scenario 2

	Key Assumptions
Scenario 2	• 7.5% waste prevention of biowaste arisings • 60% food waste capture by 2020 • 90% garden waste capture by 2020 • **Additional food waste treated by lowest social cost treatment option for each country; additional garden waste collected treated by IVC**

9.2.1 Waste Movements Resulting from Each Scenario

Table 9-2 shows the changes in the management of waste anticipated to occur under Scenario 2 relative to the baseline situation. A total of 117 million tonnes of waste is estimated to be removed from residual waste treatment facilities. Of the waste removed from these facilities, approximately 83% is assumed to be treated by some form of organic treatment method treating source-separated material. The remainder is removed from the treatment system entirely (the “waste prevention” impact).

Table 9-2: Mass Flows – Changes from Baseline, Scenario 2

	Waste movements (thousand tonnes)								
	Waste Prevention	**Landfill**	**Incineration**	**MBT**	**Composting - Food**	**Composting - Green**	**Home composting**	**AD - Food**	**AD - Green**
2013	0	0	0	0	0	0	0	0	0
2014	-697	-1,058	-1,070	-1,207	1,643	728	-8	290	-15
2015	-1,407	-2,040	-2,211	-2,440	3,284	1,456	-18	594	-31
2016	-2,124	-1,961	-3,367	-4,686	4,894	2,167	-29	909	-51
2017	-2,849	-2,830	-4,600	-5,910	6,501	2,870	-41	1,234	-72
2018	-4,299	-4,915	-7,155	-8,733	10,038	4,752	-65	1,894	-115
2019	-5,764	-6,837	-10,215	-11,043	13,461	6,572	-92	2,554	-163
2020	-7,243	-7,823	-12,775	-14,635	16,776	8,332	-122	3,217	-213
Total	*-24,383*	*-27,464*	*-41,393*	*-48,655*	*56,597*	*26,875*	*-374*	*10,692*	*-660*

The Table indicates that approximately 28 million tonnes of additional annual biowaste treatment capacity need to be in place, across the EU, by 2020 to meet the targets in Scenario 2. Since the baseline is assumed to meet Landfill Directive targets, this total capacity also represents the additional biowaste diversion beyond Landfill Directive targets. This net reduction beyond Landfill Directive targets is shown for each member state in Figure 9-1. Although the biowaste treatment systems vary between the two scenarios, this chart also applies to Scenario 2a since targets under the two scenarios are the same.

Figure 9-1: Reduction in biowaste to landfill beyond Landfill Directive targets for Scenario 2 and 2a (total 2013-2020)

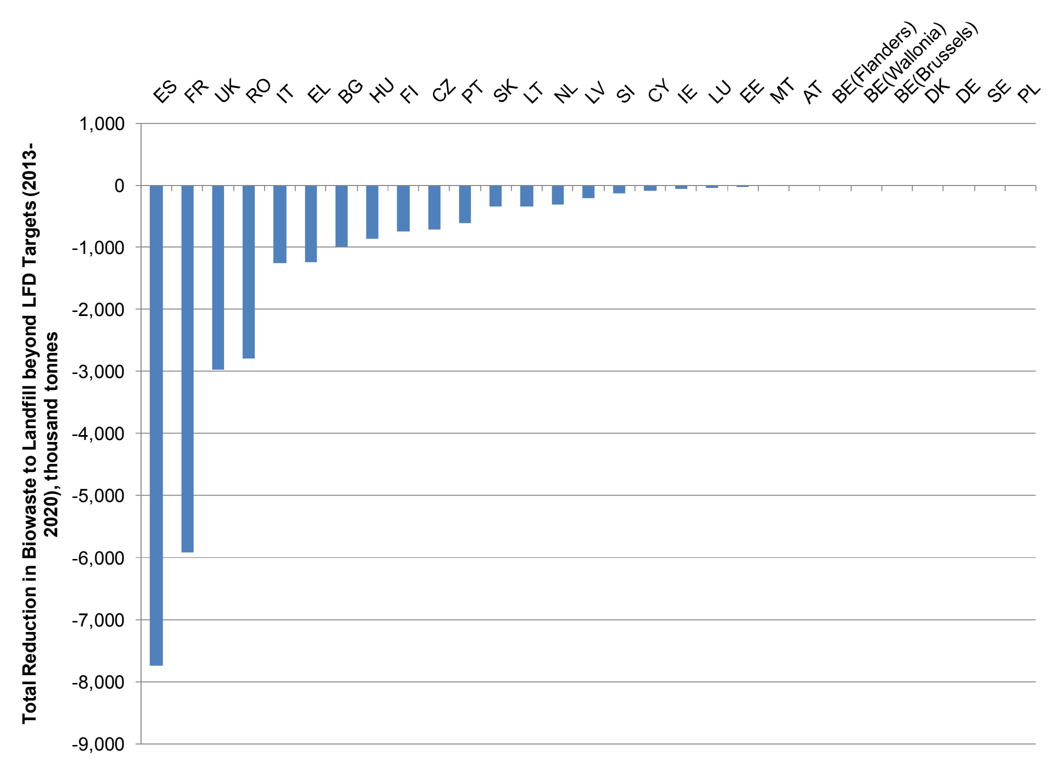
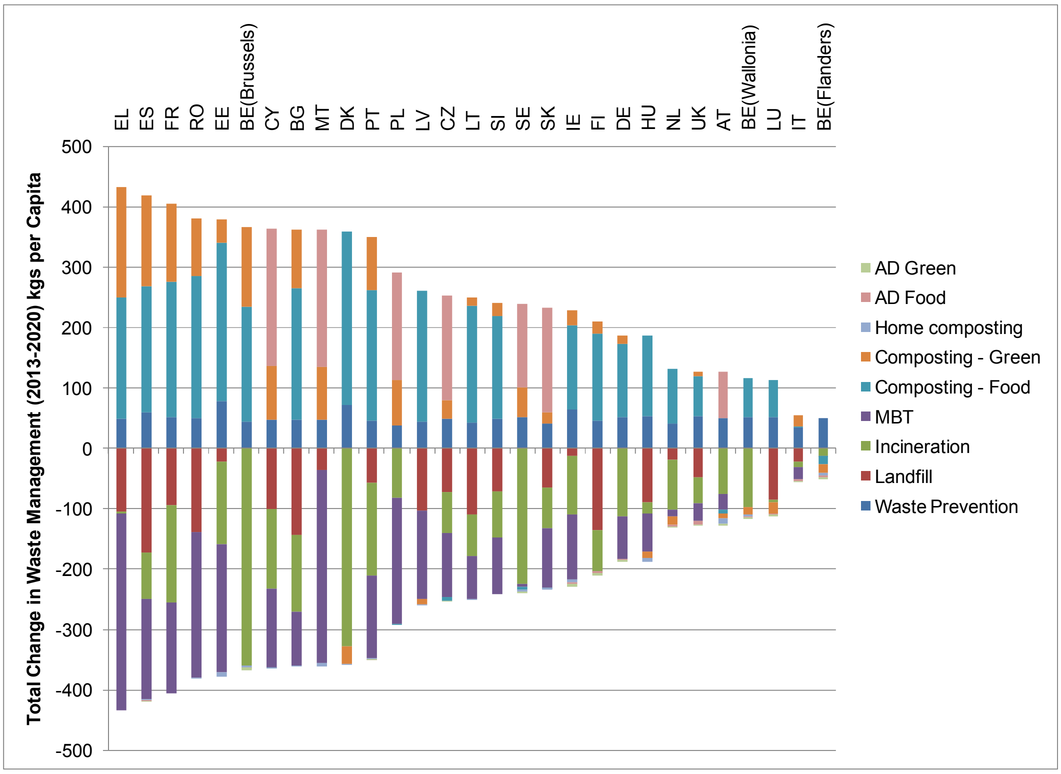


Figure 9-2 shows the change in waste management, for each country (in kgs per Capita), as a result of policy Scenario 2. The negative columns show what waste management process the waste has been diverted from. It is assumed that the relative shares of landfilling, incineration and MBT in the treatment of the waste that is not collected separately remains as in the baseline. The only exceptions to this rule are where it has been necessary to assume significant increases in residual treatment to meet the landfill directive targets in the final target year (2016 or 2020). In the instances where the additional source separation of biowaste required by the potential biowaste policy reduces the need for the additional residual treatment capacity, then we assume that this is not put in place.

The positive columns are equal to the negative, potentially implying that the net change in arisings is zero. However, the positive element also includes the effect of waste prevention, so the total waste managed *is* less overall. The magnitude of the switch to biowaste treatments is directly related to the current level of source separation, of both food and garden waste, in the baseline. In Flanders, for example, where both targets are already met, no additional collection of food or garden waste is required. This is shown in the chart. The positive element for Flanders is, as mentioned previously, only the effect of 7.5% waste prevention of biowaste arisings. The negative element indicates the quantities of waste diverted from the baseline as a result of this prevention effect. These are in the same proportions as those in the baseline.

Figure 9-2: Scenario 2 - Total Change in Waste Management (2013-2020), kgs per Capita



Luxembourg is already meeting the garden waste target, thus the only switch is to food waste treatment, in this case composting. In Scenario 2, the destination of the biowaste is determined by the net social cost of the treatments. For most countries, the lowest net social cost for food waste treatment is composting (light blue in chart), with only some showing anaerobic digestion (pink in chart) as the best performing option by this measure. Where countries have composting as the preferred food waste treatment method all additionally collection garden waste is assumed to be treated co-mingled in suitable biowaste facilities (such as in-vessel composters). Where AD is preferred, the additionally collected green waste is assumed to be treated in lower cost open air windrow systems.

In countries where there is little source separation, of either food or green waste, the change in waste management to meet the proposed targets set out for Scenario 2 is significant. In Greece, for example, a significant diversion of biowaste from baseline residual treatments is required.

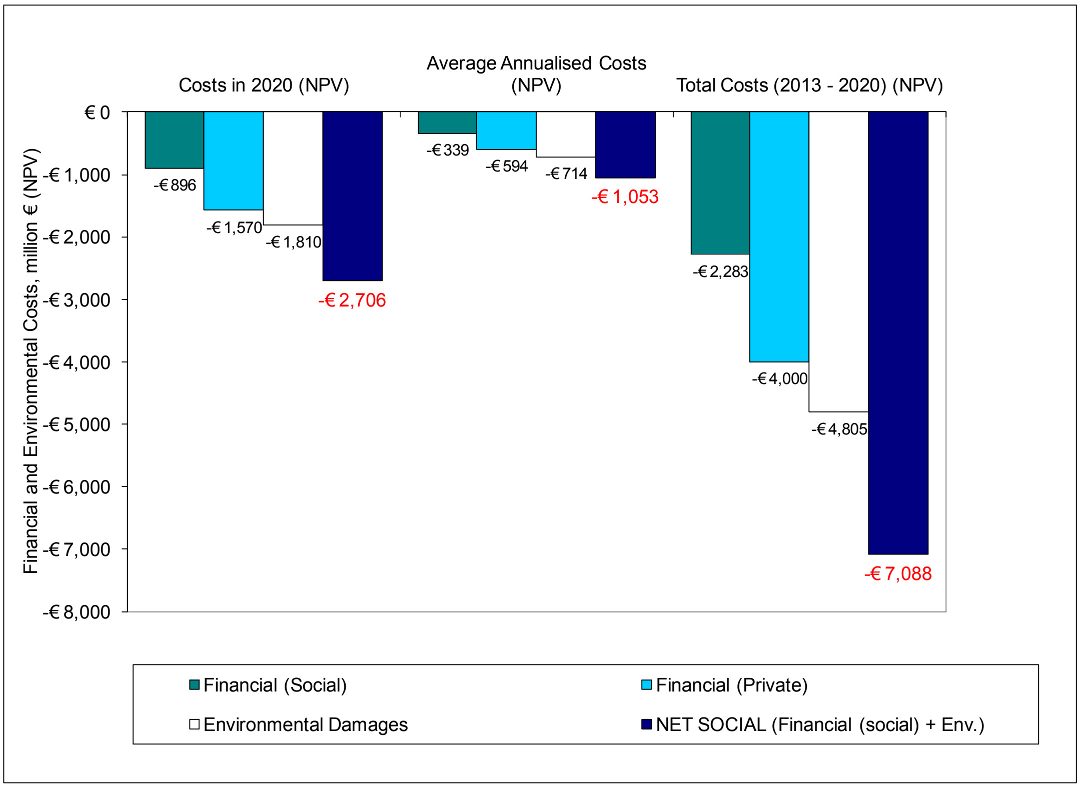
The key point to draw out from this chart is that the *main factor* driving the change in quantity, of separately collected biowaste, required to meet the targets (which varies from none (Flanders), through to some (Czech Republic), and to significant quantities (Spain or Greece) is directly related to *the level of source separation in the baseline*. This is not surprising, but it is worth pointing out that other factors, such as the mix of existing residual waste treatments, were not considered a limiting factor. The tonnage changes presented here will have one of the most significant factors affecting the net greenhouse gas emissions and costs in the sections that follow.

9.2.2 Financial and Environmental Costs of Scenario 2

Figure 9-3 presents the total financial and environmental costs of Scenario 2 for all 27 Member States combined, over the period 2013-2020. The graph indicates that Scenario 2 results in a significant net benefit to society, calculated from adding together the environmental damages and the financial cost under the social metric. **Scenario 2 gives the greatest benefit of all the Scenarios modelled**, resulting in a **net benefit to society of €7,088 million over the period 2013-2020**. The chart also shows the data for the situation in 2020 and the average annualised costs between 2013 and 2020. This shows that the greatest benefits from the policy accrue towards the end of the time period modelled in this study.

The net social benefits relate partly to cost savings, and partly to environmental benefits (financial cost under social metric plus environmental cost). The environmental benefits would be higher if the analysis suggested that the preferred option for treating biowaste, from the perspective of society, was anaerobic digestion in a greater number of countries. On the other hand, the financial savings would be lower, so that the net benefit to society would be slightly reduced, but with a greater proportion of the benefit being derived from environmental benefits rather than financial savings. This will be made clear through modelling of Scenario 2a (Section 9.3).

Figure 9-3: Total Financial and Environmental Costs for the EU-27, Scenario 2



Note: negative figures imply a net saving

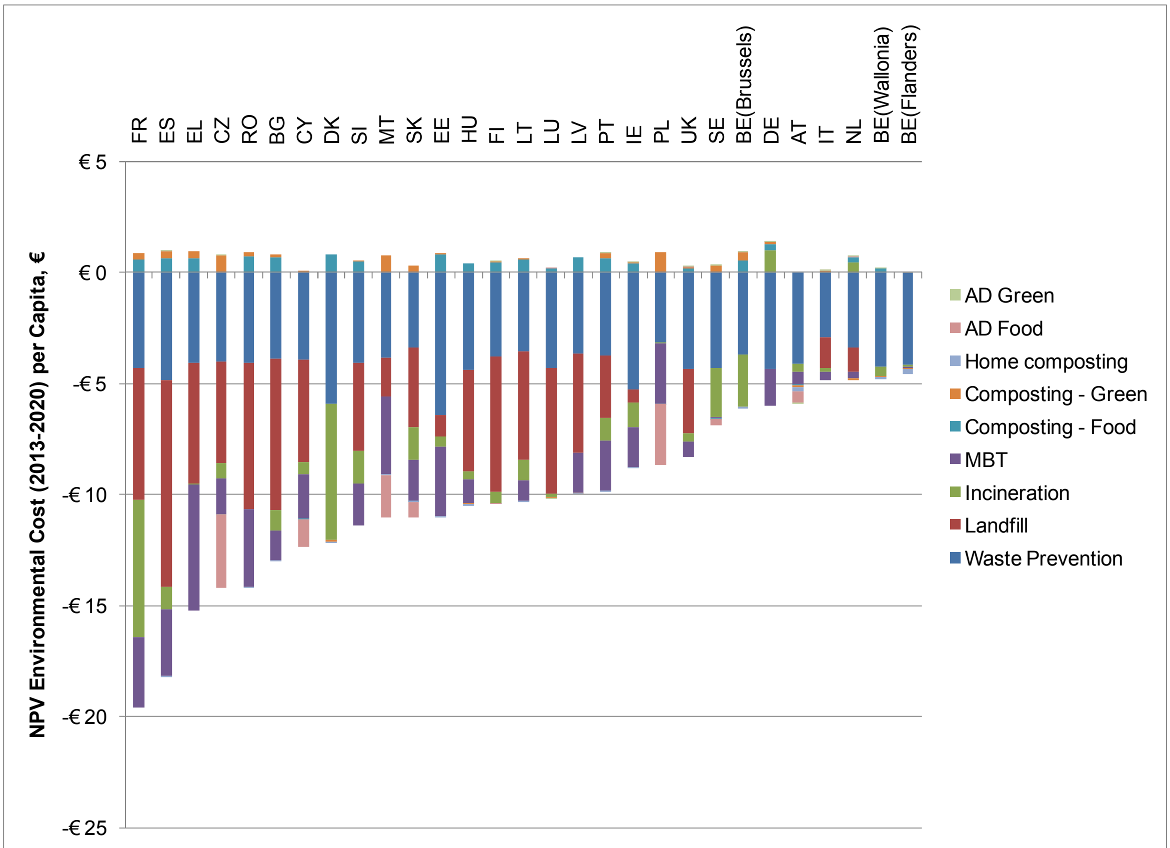
To explain where the net cost to society has been derived from the component parts of financial costs and environmental costs have to be shown for each country. Firstly, a discussion of the environmental costs is undertaken, followed by a discussion of financial costs (under the social metric).

Figure 9-4, below shows what the net environmental cost figure is comprised of, by country. It is clear that there is significant environmental benefit from the majority of the changes made in waste management required to meet the targets.

The positive elements represent a *cost* to society. These are small, but some consideration needs to be given. In Germany and the Netherlands, there is actually a cost to society from switching biowaste from incineration to source separated collection and treatment. This is due to the baseline assumptions in relation to pollution abatement and avoided source of energy generation. In Germany, for example, selective catalytic reduction (SCR) is common on incinerators, and under electricity only mode, the process generates a significant quantity of electricity that offsets the generation by the national, mainly coal based, power system. This coal based system would be producing a large quantity of NOx, and as the damage costs in Germany are also high, the benefit, in terms of air quality, is significant.

An increase in any composting processes, but to a greater extent open air windrow will results in an increase in environmental costs. This relates directly to the level of ammonia emitted from the processes, and the relatively high damage costs attributed to these emissions. It should be noted that the damage costs attributed to emissions of ammonia may be over-stated. The damage costs from the CAFE study appear to relate mainly to significant point sources, and are linked to the development of secondary particulate matter. If, as may be the case in some areas where composting takes place, the presence of SOx and NOx (and not ammonia) is the limiting factor in secondary particulate formation, then ammonia-related damages would be expected to be much lower.

Figure 9-4: Breakdown of Environmental Damage Costs, by Treatment and Country, NPV (2013-2020) per Capita, €



Note: negative figures imply a net saving

All of the negative elements indicate where the environmental benefits are generated. There is a significant benefit for all countries from waste prevention. Especially for those where little additional collection of biowaste is required (Flanders, Austria and Germany). Where the lowest social cost biowaste treatment option is AD, there is a benefit (Poland, Austria, Czech Republic and Sweden, for example). The level of benefit relates to the quantity of waste switched and the preferred AD process (See Section 7.7).

As well as the stark waste prevention benefits, significant levels of environmental benefit, per capita, also occur when biowaste is diverted from landfill,. This is shown clearly in the chart. The countries for which this occurs are Spain, France, Romania and Bulgaria (amongst others). Additional benefit will also be derived from switching away from other residual treatments, existing, or offset in the baseline, through the introduction of the policy. Apart from the two examples above, for all processes, in every other country, there is an environmental benefit from switching waste away from residual treatment technologies into source separated biowaste treatments. Of fundamental importance are the benefits associated with waste prevention. Because waste prevention is associated with 4.5tonnes of CO_2 reduction per tonne prevented, the associated environmental benefits in each member state are significant, whether nor not the policy leads to other sizable shifts in material between different treatment systems.[110]

As can be seen from the chart the greatest environmental benefits, per capita, occur from countries such as France, Greece, Spain, Czech Republic, Romania and Bulgaria. The following two factors are the most significant in determining these levels of environmental benefit:

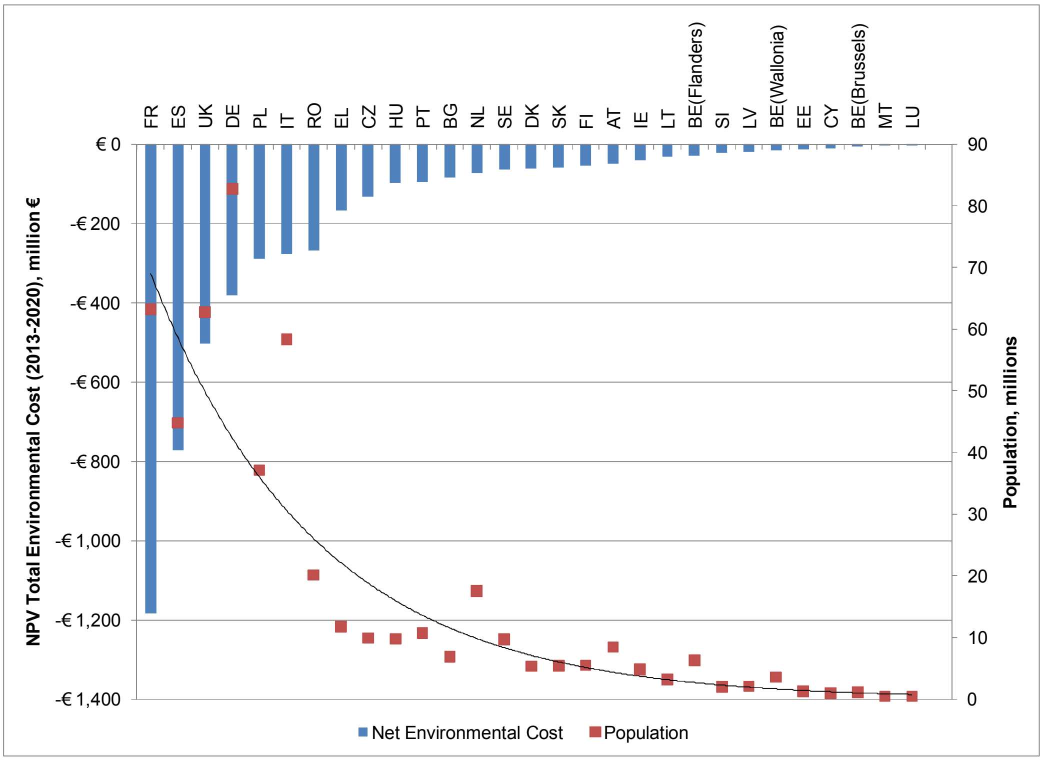
(i) The low levels of source separated biowaste collection in the baseline; and

(ii) The level of landfilling in the baseline and resultant switch from this disposal method.

When total cost figures are considered, as shown by Figure 9-5, the larger countries for which the levels of benefit per capita were great are also the most significant (France and Spain). Smaller countries (Romania, for example) still provide high levels of benefit. The results, however, are much more strongly determined by the population of the country, and (in turn) the waste prevention and diversion associated with the policy – a higher populous leading greater benefits. High levels of environmental benefit are achievable in the UK despite the benefit per capita being relatively small compared with other Member States. Following the same logic the smaller countries, by population, (Malta and Luxembourg) generate the smallest levels of benefit. This is shown clearly in the chart. In fact it does appear as though there is quite a strong correlation between the environmental benefit and the size of the population, though of course, there is no deterministic link between these two factors (with Germany, for example, showing a notable bucking of this trend owing to its strong waste management performance in the Baseline).

[110] Barthel, M, “The Importance of Tackling Food Waste”, WRAP, http://www.wrap.org.uk/downloads/Mark_Barthel_-_Introduction_context.4b08ccfa.5939.pdf

Figure 9-5: NPV Total Environmental Costs (2013-2020), million € / Population, millions



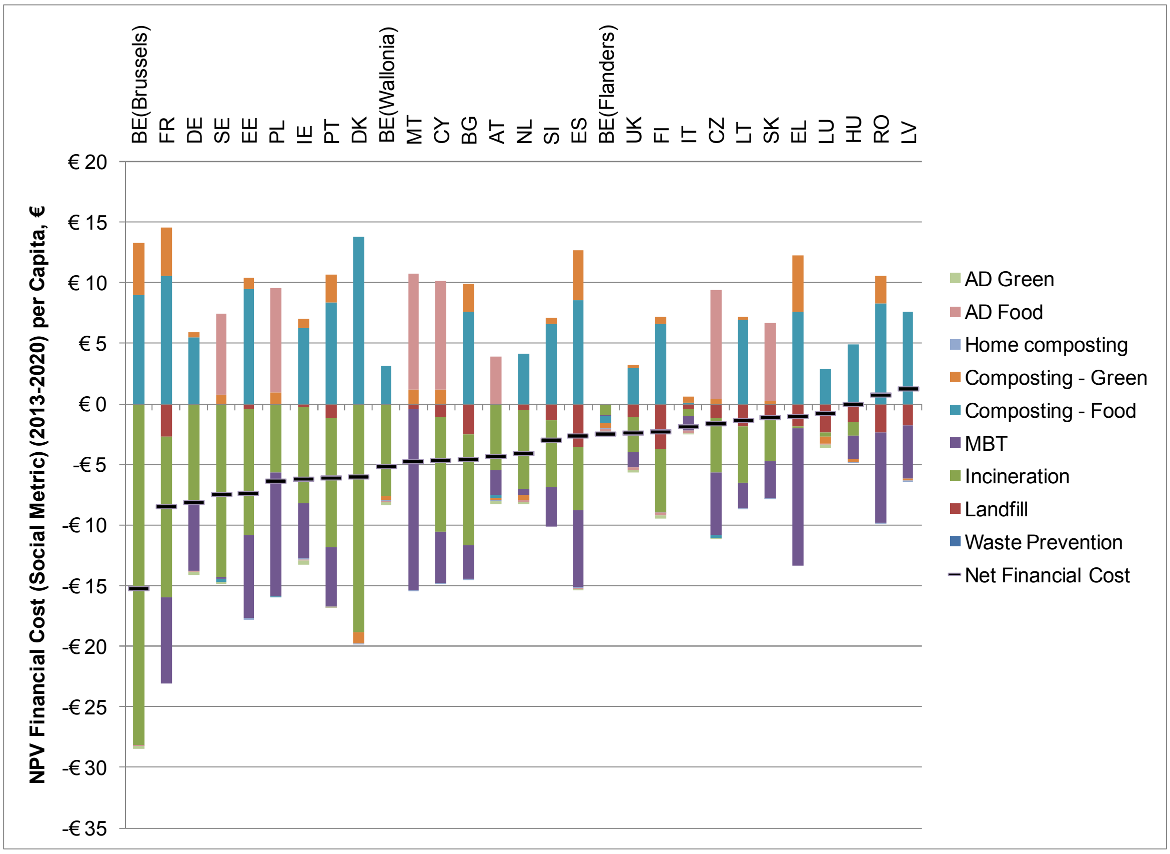
Therefore, the key factor influencing the total environmental benefit across the EU appears to be the size of the country and the quantities of waste generated within it. However, clearly the benefit per capita (shown in Figure 9-4) has some bearing also.

The second element to consider when discussing the net cost to society is the financial cost elements. Figure 9-6 shows the financial savings, attributable to the reduction in need for residual waste treatment, and the financial costs for treating the source separated biowaste. Note that the assumption has been made that the net change in collection costs will be zero (see Section 7.4).

For all countries, except Romania and Latvia, there is a financial saving from the policy. This is shown by the negative Net Financial Cost indicators on the chart. It should be re-iterated here that the Baselines imply that all countries meet Landfill Directive targets, so that no country is assumed to simply continue landfilling.

The cost of biowaste treatments is clearly related to what is the preferred option for each country. Sweden, Poland and Malta, for example, show costs for treatment of food waste in AD, whereas Brussels shows costs for food *and* garden waste treatment in composting facilities, as composting is the preferred option. The countries for which the cost of biowaste treatment per capita is higher, correlates well to the additional quantity of material collected for biowaste treatment needed to meet the targets (see Figure 9-2). Effectively, if more biowaste needs to be separately collected for biowaste treatment, the cost per capita is higher.

Figure 9-6: Breakdown of Financial Costs (Social Cost Metric), by Treatment and Country, NPV (2013-2020) per Capita, € (ranked by Net Financial Cost)

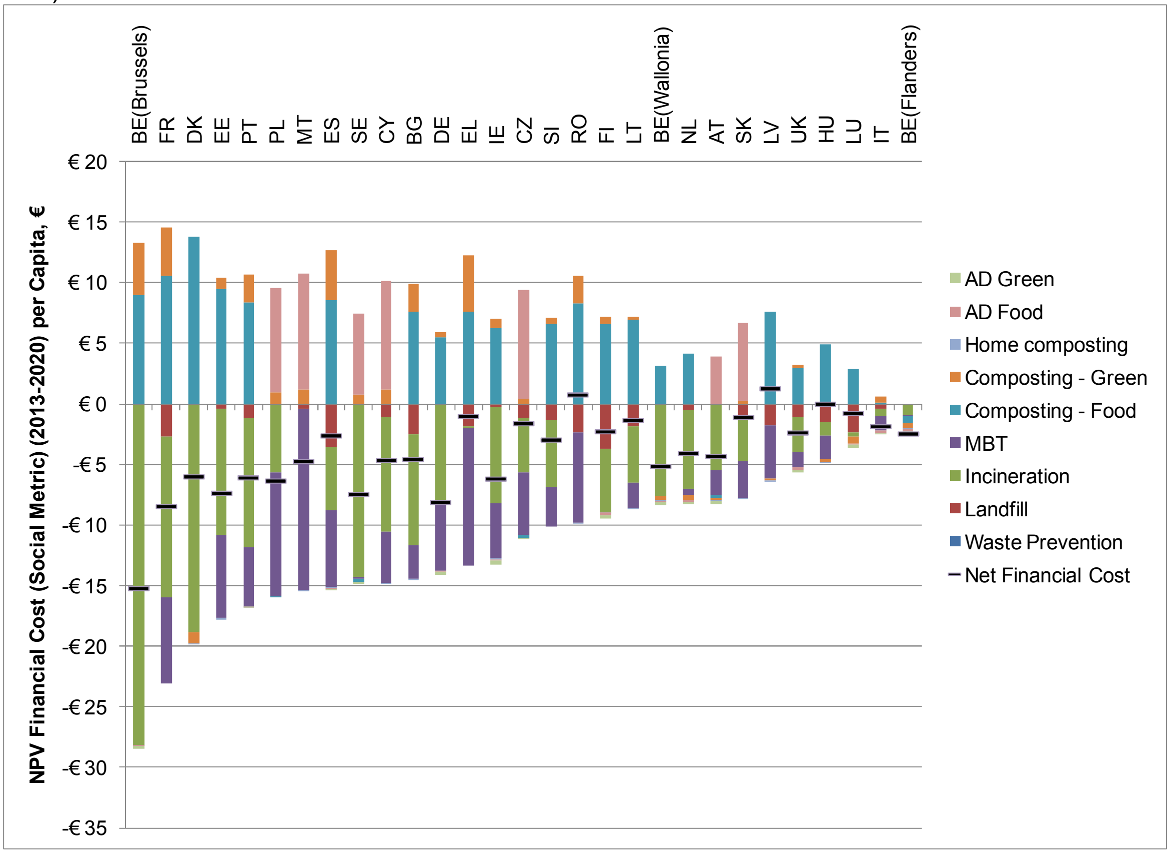


The financial savings from the policy occur because the avoided cost of residual waste treatment /disposal is greater than the cost of treating separately collected biowaste. In countries where biowaste treatment is comparable to that for residual waste (Hungary, Slovakia, Lithuania and the Czech Republic for example), the financial savings are low. However, when the cost of residual treatment is higher (Spain, Denmark, Estonia, France and Brussels for example), the financial savings from the policy increase.

To understand what factors lead to different Net Financial Costs, and specifically the financial savings in each country, it is helpful to discuss countries in relation to the avoided residual waste treatment cost. Figure 9-7 below shows the countries ranked from highest avoided cost (Brussels) to lowest (Flanders). For Flanders, the only cost saving is as a result of waste prevention. The diversion to biowaste treatment is zero.

To understand one of the key factors that determines the avoided cost of disposal, for each country, the administrative regions of Belgium can be used as an example. Figure 9-7 shows that, in both Wallonia and Brussels, biowaste is only diverted from incineration to treatment of separately collected biowaste, hence they can be used for comparison. The financial cost of incineration, per tonne, in Wallonia and Brussels is € 99.6 and € 99.8 respectively. Therefore, the significant difference in net financial costs is *not* due to the relative *cost* per tonne of avoiding disposal, it is due to the *quantity* of waste diverted to biowaste treatments. This can be seen clearly in Figure 9-2 also.

Figure 9-7: Breakdown of Financial Costs (Social Cost Metric), by Treatment and Country, NPV (2013-2020) per Capita, € (ranked by avoided disposal / treatment cost)



By the same measure, Brussels and Romania are comparable, but the avoided disposal / treatment costs are around 28 € and 10 € respectively. A significant difference. Thus, in this case, the difference in costs is not due to the quantity of waste diverted to biowaste treatments, but the type, and related cost, of the treatment being offset. Clearly, in Romania, the mix of residual treatments is different (landfill and MBT vis-à-vis incineration). Landfill and MBT are cheaper in Romania, but the underlying reason is drawn out when the same technology is compared. The annualised cost of incineration in Brussels and Romania is € 99.8 and € 80.9 respectively. The differences are explained in detail in Section 7.8, however, the key factors are the wholesale price of electricity and the relative labour and capital costs of infrastructure.

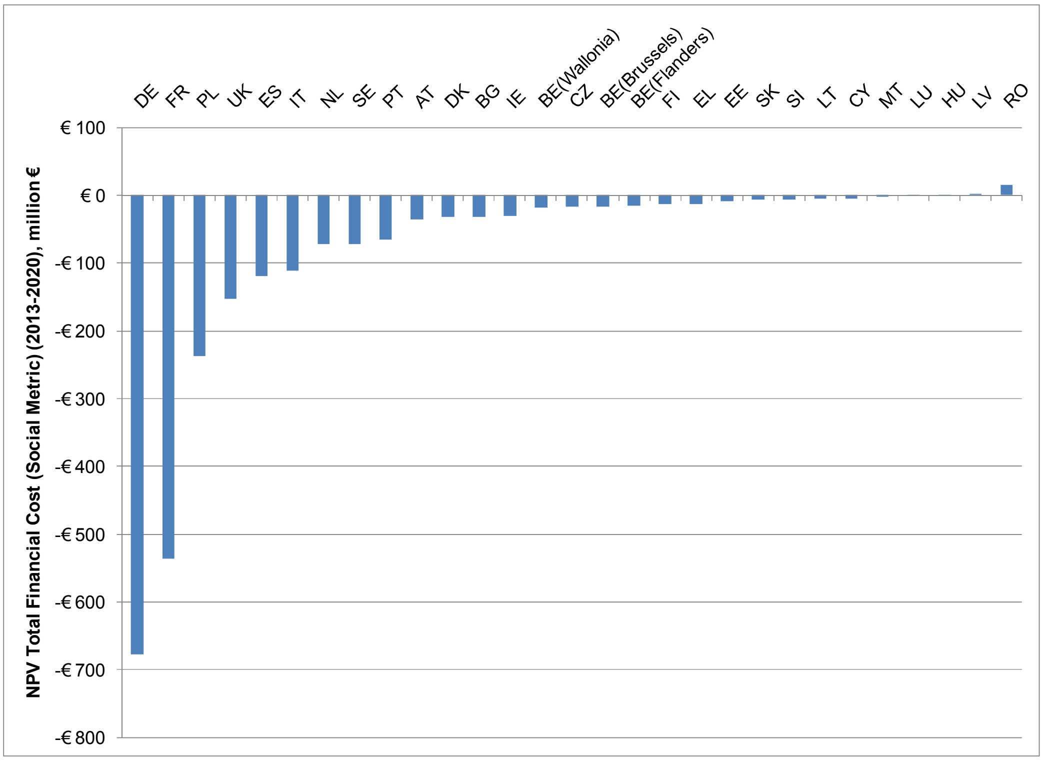
So the main factors that affect the financial benefit, per capita, of the policy for each country are:

(i) The change in quantity of separately collected biowaste; and

(ii) The costs of avoided treatment / disposal.

When the total financial costs are considered a similar pattern to Environmental costs emerges. The countries that generate a greater saving are larger, by population, and therefore generate more waste. More waste is moved and thus the savings are greater.

Figure 9-8: NPV Total Financial Costs (Social Metric) (2013-2020), million €



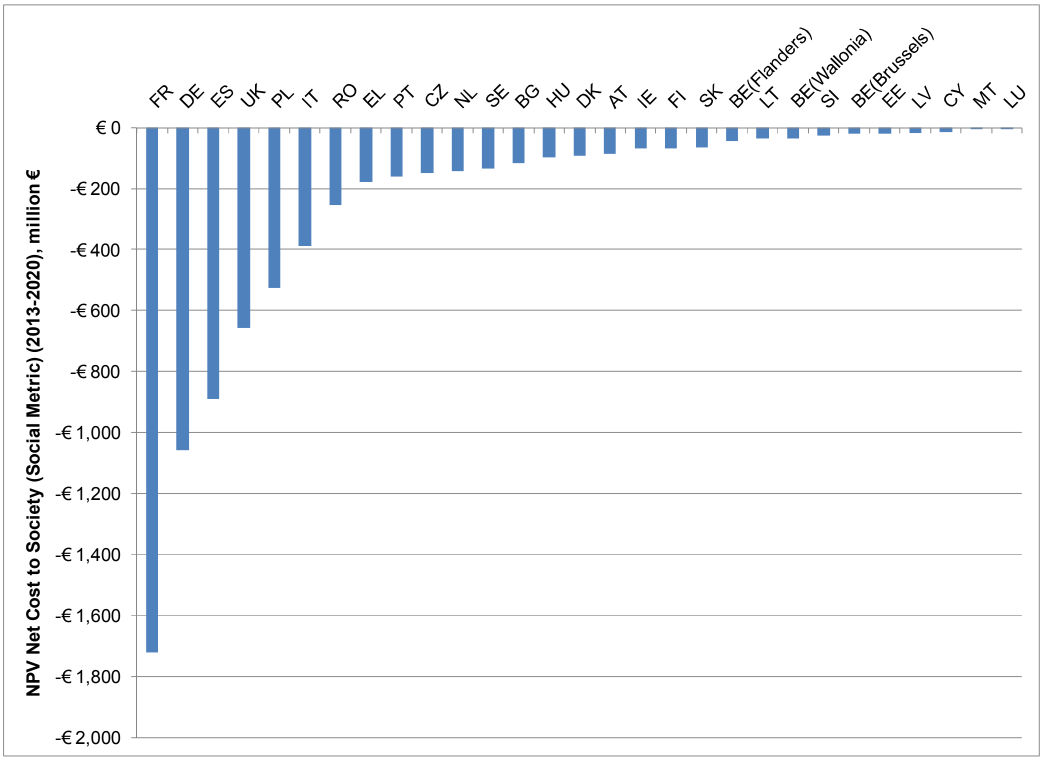
When both financial and environmental costs are considered the Net Cost to Society can be calculated. Based upon the analysis presented above it is clear that the most significant factors affecting the Net Cost to Society, for each Member State, are:

(i) The amount of source separated biowaste treatment in the baseline; and

(ii) The size of the country and household waste generated within it.

The breakdown of Total Net Cost to Society (-€ 7,088), shown in Figure 9-3, by country, is shown below in Figure 9-9.

Figure 9-9: Scenario 2 – NPV Net Cost to Society (Social Metric) (2013-2020), million €



9.2.3 Changes in Greenhouse Gas Emissions, Scenario 2

Greenhouse gas (GHG) emissions for each type of treatment will vary for each country, depending upon the fuel mix used for electricity and heat generation. Impacts calculated on the basis of one tonne treated by each facility will be particularly variable for those facilities that are assumed to generate or use large quantities of energy, such as incineration facilities or the AD facilities generating energy on-site. As an example, Table 9-3 shows the GHG emissions for treating one tonne of waste by each of the treatment methods for the Czech Republic. **Impacts are shown both inclusive and exclusive of the biogenic CO_2 emissions,** which are typically excluded under analyses following a life-cycle approach to calculation.

Table 9-3: Example Unit GHG Emissions for the Czech Republic

	GHG emissions from treating one tonne of material	
	Including biogenic CO_2 eq	**Excluding biogenic CO_2 eq**
Waste Prevention	-4.50	-4.50
Landfill	1.47	1.09
Incineration	0.36	-0.23
MBT	0.70	0.14
Composting – Food	0.49	0.11
Composting – Green	0.43	0.01
AD - Food	0.49	0.11
AD - Green	0.34	-0.04
Notes: The incineration facility is assumed to generate only electricity in this example. MBT is stabilisation with output to landfill. Impacts associated with composting green waste		

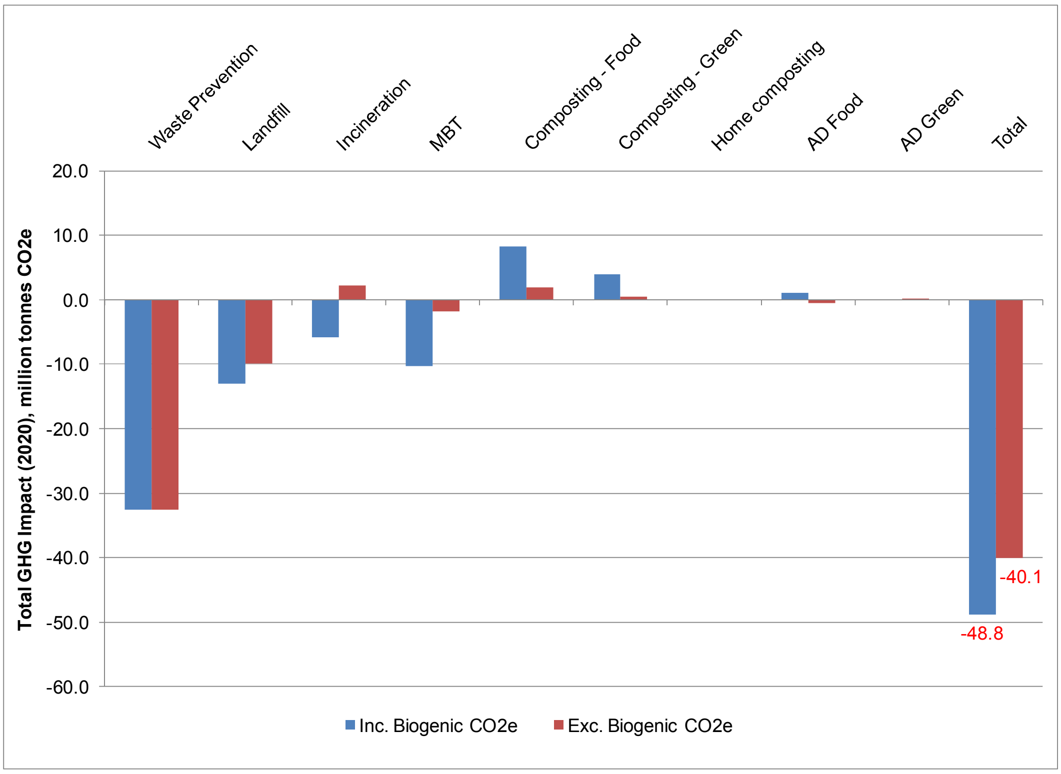
assume the waste is treated in a windrow facility.

Figure 9-10 presents the greenhouse gas impacts anticipated to follow from the changes in treatment methods that result from Scenario 2. Results are shown for the year 2020, by which time it is anticipated that the full impact of the policy will be seen. The graphic should be read as showing the changes in emissions from each management route in the scenario. So, for example, where a negative bar is shown against landfill, this does not mean that landfilling implies a net GHG saving. It shows how the change in quantity landfilled translates into changes in GHG emissions.

Scenario 2 shows a significant reduction in overall GHG emissions. This is true in both the case where emissions of CO_2 eq from biogenic sources are included, and where they are excluded. Note that our estimate of benefits from waste prevention is the same in both cases, and since this appears to be a value for the case where CO_2 eq from biogenic sources are excluded, the effect of waste prevention might be underestimated in the case where CO_2 eq from biogenic sources is included in the calculation (the waste prevention benefits could increase). **The main effects come from waste prevention, and avoided landfilling and MBT.** Note that the use of biowaste treatments gives rise to net emissions of GHGs in both cases (much reduced, in the case where CO_2 eq from biogenic sources is excluded from the calculation, due to the majority of emissions being biogenic in nature).

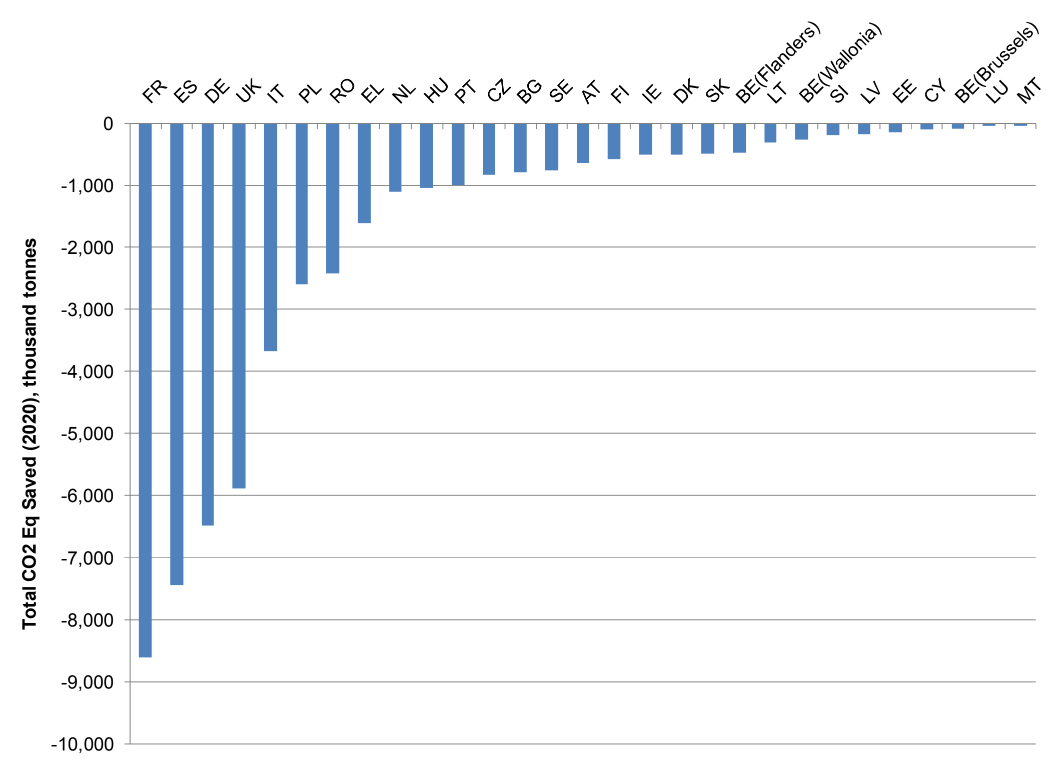
The total benefit is a reduction in 48.8 million tonne CO_2 eq where CO_2 eq from biogenic sources is included in the calculation, and 40.1 million tonnes where it is not.

Figure 9-10: Total Greenhouse Gas Implications of Scenario 2 for the Year 2020 (Including & Excluding Biogenic CO_2 eq Impacts)



The results per country show a similar pattern. In overall terms the policy will deliver a net GHG savings for each Member State. This is shown in Figure 9-11 below.

Figure 9-11: Total Greenhouse Gas Implications of Scenario 2 for the Year 2020 (Including Biogenic CO_2 eq Impacts) for each Country



9.3 IMPACT ASSESSMENT – SCENARIO 2a

The following sections identify the changes in costs and benefits, as well as greenhouse gas impacts, of Scenario 2a. Table 9-4 summarises the scenario considered within the analysis.

Table 9-4: Key Assumptions Underpinning Scenario 2a

	Key Assumptions
Scenario 2a	• 7.5% waste prevention of biowaste arisings • 60% food waste capture by 2020 • 90% garden waste capture by 2020 • **Food waste treated by option with best greenhouse gas emissions outcome for each country**

The following table indicates, for each country, which biowaste treatment abates the greatest quantity of GHGs.

Lowest GHG Impact Biowaste Treatment	Country
AD: compressed biogas used in vehicles	AT, BE(Flanders), BE(Wallonia), BE(Brussels), BG, FI, FR, HU, LV, LT, LU, SK, SI, ES, SE
AD: on-site biogas use + CHP	CY, CZ, DK, EE, DE, EL, IE, IT, MT, NL, PL, PT, RO, UK

The mass flows are effectively the same as under Scenario 2. However, there are some differences relative to Scenario 2, and these are shown in Table 9-5. . The net additional biowaste diversion achieved beyond landfill directive targets remains the same as in Scenario 2, as previously shown for each member state in Figure 9-1.

Table 9-5: Difference in Mass Flows Between Scenarios 2 and 2a

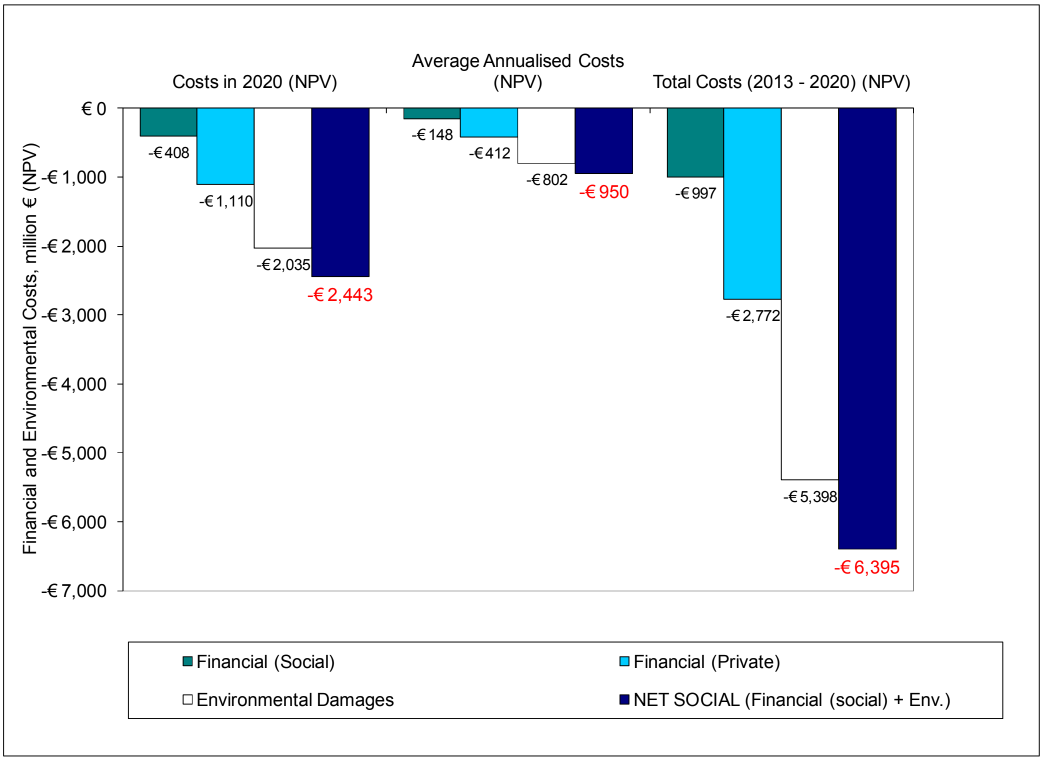
	Waste movements (thousand tonnes)	
	Compost	**Anaerobic Digestion**
2013	0	0
2014	-1,690	1,690
2015	-3,384	3,384
2016	-5,051	5,051
2017	-6,720	6,720
2018	-10,382	10,382
2019	-13,939	13,939
2020	-17,399	17,399

9.3.1 Financial and Environmental Costs of Scenario 2a

Figure 9-12 below presents the total financial and environmental costs of Scenario 2a for all 27 Member States combined, over the period 2013-2020. The graph indicates that Scenario 2a results in a significant net benefit to society, calculated from adding together the environmental damages and the financial cost under the social metric. **Scenario 2a gives, as would be expected, a lower net social benefit relative to Scenario 2, the difference being 10%.**

The key difference relative to Scenario 2 is that the net social benefits derive not from the savings in financial costs, but the higher savings in environmental damages (around € 5.4 billion in Net Present Value terms for the 2013-2020 period). Evidently, some of these relate to waste prevention effects, but a significant benefit comes from the choice of treatment for food wastes.

Figure 9-12: Total Financial and Environmental Costs for the EU-27, Scenario 2a



The pattern of financial and environmental costs is similar to that under Scenario 2. Figure 9-3 and Figure 9-12 would look similar, but with all countries showing changes in costs associated with AD, rather than composting. However, as alluded to above, the key differences are:

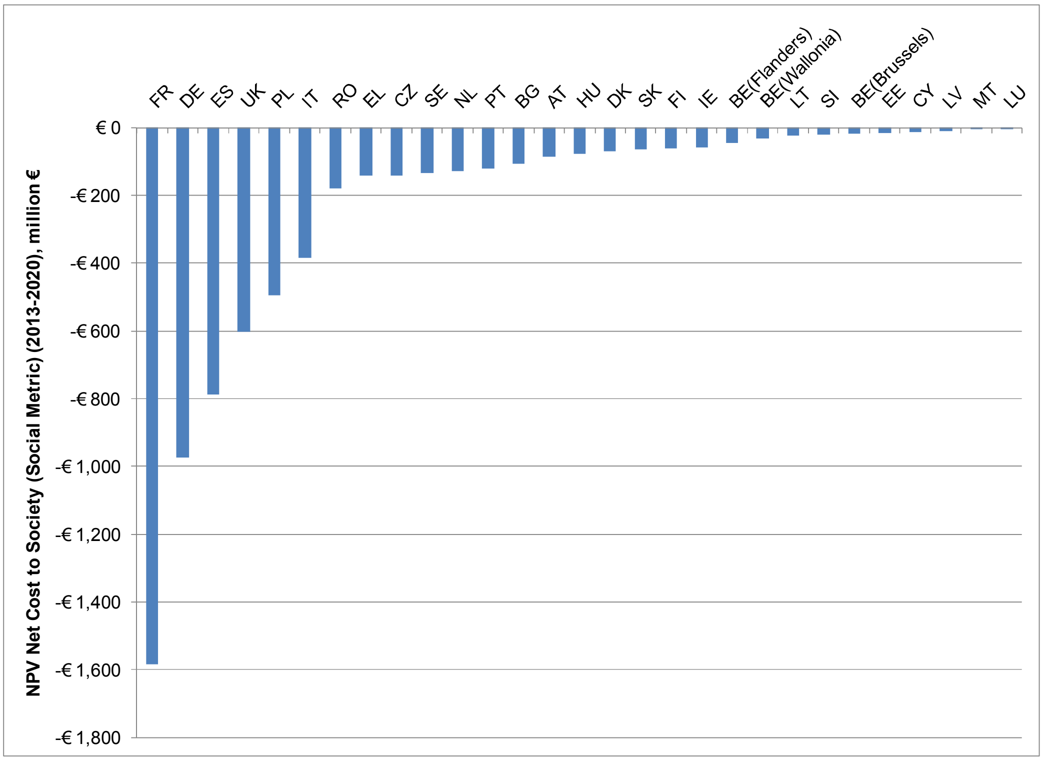
(i) Environmental costs decrease as more GHGs are abated through using AD technologies; and

(ii) Financial costs increase, as these technologies are more expensive.

The environmental benefit from requiring food waste to be managed with AD increases by 12% overall. There is, however, a greater loss in financial savings as a result of the policy – around 50%. However, it is important to remember that there is still a financial *saving* under the Scenario. Hence the message is that there is a greater environmental *benefit* from Scenario 2a **and** a financial cost *saving*.

The breakdown of Total Net Cost to Society, shown in Figure 9-12 (-€ 6,395), by country is shown below in Figure 9-13.

Figure 9-13: Scenario 2a – NPV Net Cost to Society (Social Metric) (2013-2020), million €



There are few small changes in the relative performance of countries, compared with Scenario 2 (see Figure 9-9). However, the key factors that affect the Net Cost to Society are still the same. These being:

(i) The amount of source separated biowaste treatment in the baseline; and

(ii) The size of the country and household waste generated within it.

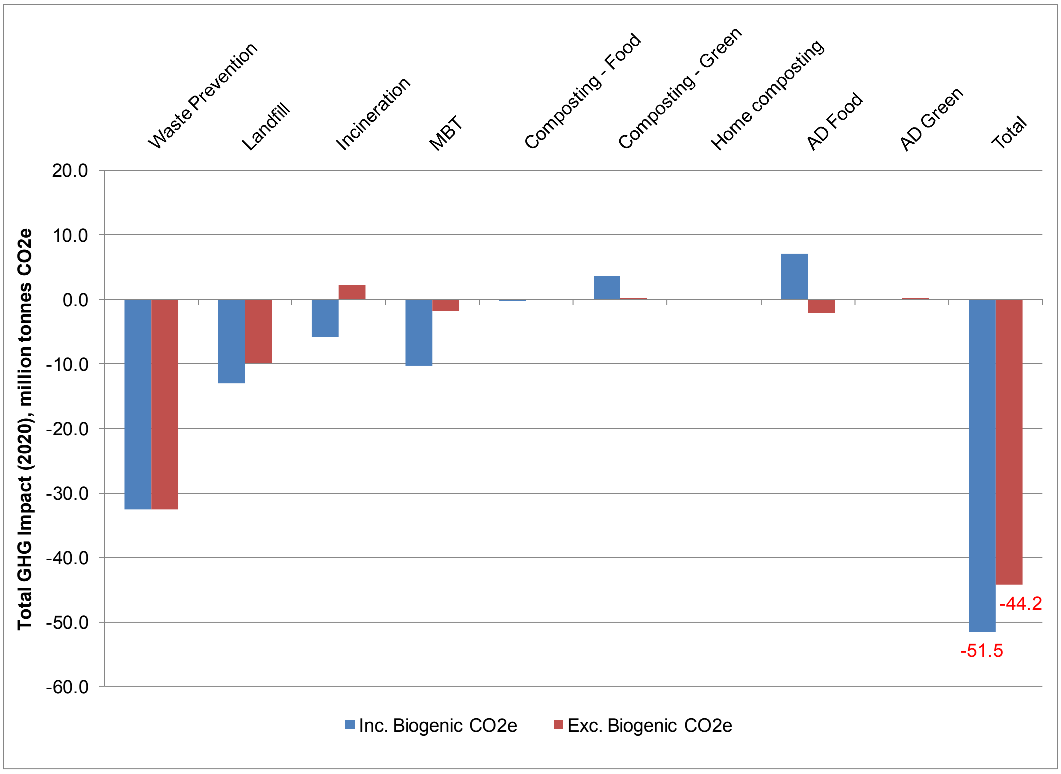
9.3.2 Changes in Greenhouse Gas Emissions, Scenario 2a

Figure 9-14 presents the greenhouse gas impacts anticipated to follow from the changes in treatment methods that result from Scenario 2a. Results are shown for the year 2020, by which time it is anticipated that the full impact of the policy will be seen. The results show the changes in GHG emissions in million tonnes of CO_2 equivalent for the EU-27 in this year. The graphics should be read as showing the changes in emissions from each management route in the scenario. So, for example, where a negative bar is shown against landfill, this does not mean that landfilling implies a net GHG saving. It shows how the change in quantity landfilled translates into changes in GHG emissions.

Scenario 2a shows a reduction in overall GHG emissions that is even more significant than under Scenario 2. This is true both in the case where emissions of CO_2 from biogenic sources are included, and where they are excluded. The greater reduction is associated with the choice of biowaste treatment option.

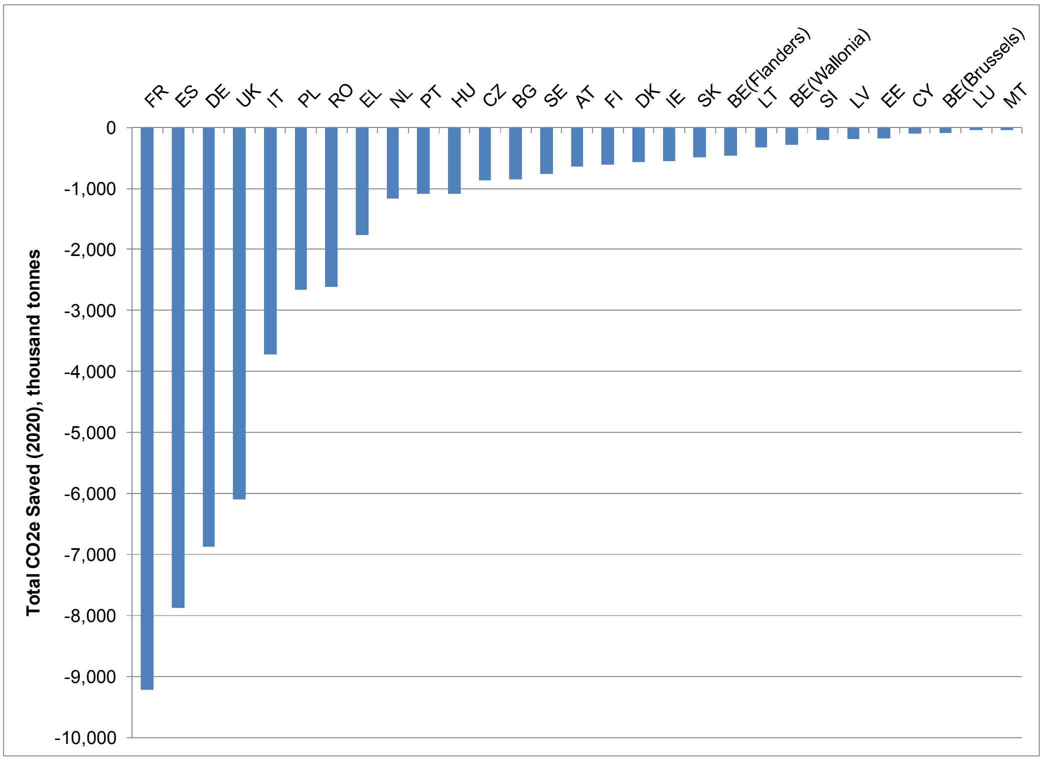
The total benefit is a reduction in 51.5 million tonne CO_2 eq where CO_2 eq from biogenic sources is included in the calculation, and 44.2 million tonnes where it is not. The difference between Scenario 2 and 2a is not especially large in relative terms (2.6 million tonnes and 4.1 million tonnes where CO_2 eq from biogenic source is included, and excluded, respectively). However, if the impact from biowaste treatment alone is considered, GHG emissions, with and without biogenic CO_2 eq included, decrease by 20% and 210% respectively.

Figure 9-14: Total Greenhouse Gas Implications of Scenario 2a for the Year 2020 (Including & Excluding Biogenic CO_2 eq Impacts)



The results per country show a similar pattern. In overall terms the policy will deliver a net GHG savings for each Member State. This is shown in Figure 9-15. As with Net Cost to Society, the amount of GHGs abated is mainly dependent upon how much additional biowaste is to be separately collected, and treated, and the size of the country.

Figure 9-15: Total Greenhouse Gas Implications of Scenario 2a for the Year 2020 (Including Biogenic CO_2 eq Impacts) for each Country



10 Third policy scenario: low recycling

10.1 Definition and methodology

The third policy scenario is determined by, on the one hand, general assumptions that apply to every policy scenario (see Section 6.2) and, on the other hand, specific assumptions on prevention and recycling targets that will be met in this policy scenario 3.

We consider two variants to this policy scenario, which differ in the assumptions concerning the treatment of separately collected food waste.

10.2 Scenario 3

The following sections identify the changes in costs and benefits, as well as greenhouse gas impacts, of Scenario 3. Table 10-1 summarises the scenario considered within the analysis.

Table 10-1: Key Assumptions Underpinning Scenario 3

	Key Assumptions
Scenario 3	• No waste prevention • All countries below currently the European target of 36.5% biowaste collection are assumed to reach this target by 2020. This is achieved by increasing garden waste collections up to the maximum of either the baseline situation or 70% of garden arisings, and any further diversion required is assumed to occur through food waste collection. • Countries above the 36.5% target continue as per the baseline collection • **Additional food waste collected separately treated by lowest social cost treatment option for each country**

Unlike Scenarios 2 and 2a, there is **no waste prevention** element assumed in Scenario 3. Under this scenario, relative to the baseline, approximately 32 million tonnes of waste is removed from residual waste treatment, over the period 2013 to 2020, to be treated using organic treatment facilities. These changes in material flows are shown in Table 10-2.

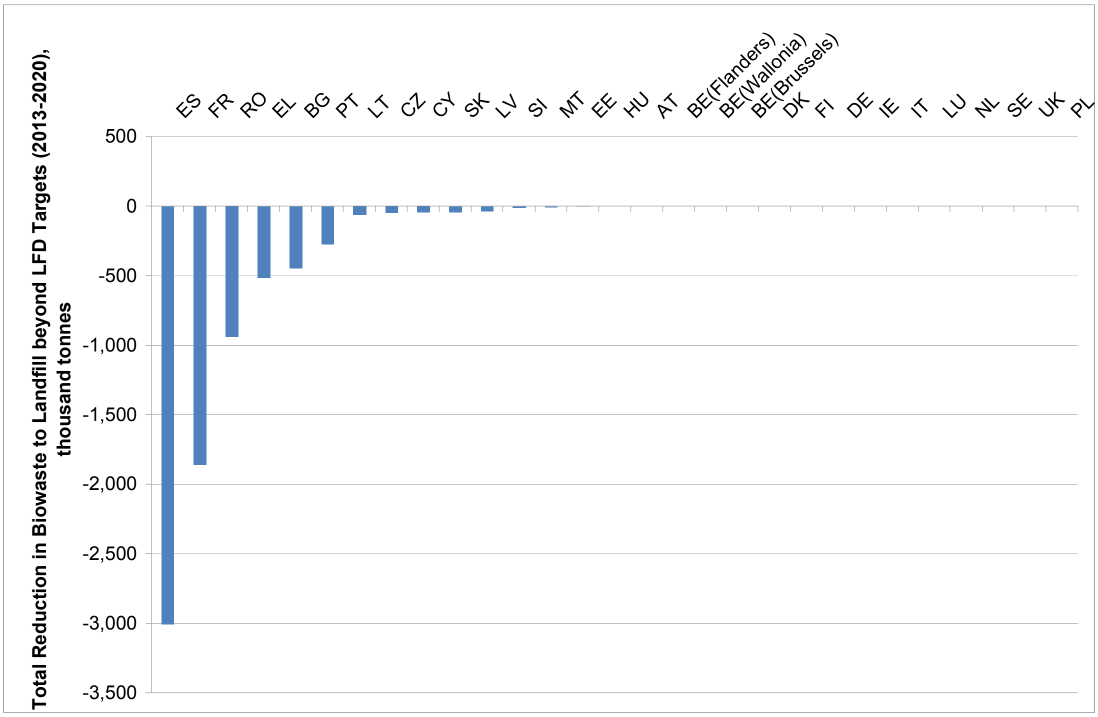
Table 10-2: Mass Flows – Scenario 3

	Waste movements (thousand tonnes)								
	Waste Prevention	**Landfill**	**Incineration**	**MBT**	**Composting - Food**	**Composting - Green**	**Home composting**	**AD - Food**	**AD - Green**
2013	0	0	0	0	0	0	0	0	0
2014	0	-521	-317	-592	358	981	0	92	0
2015	0	-881	-581	-1,022	637	1,665	0	182	0
2016	0	-674	-846	-2,021	913	2,356	0	272	0

2017	0	-991	-1,155	-2,470	1,192	3,056	0	369	0
2018	0	-1,319	-1,436	-2,955	1,471	3,768	0	472	0
2019	0	-1,635	-2,225	-2,959	1,750	4,488	0	581	0
2020	0	-1,262	-2,521	-4,161	2,029	5,218	0	697	0
Total	*0*	*-7,284*	*-9,081*	*-16,181*	*8,350*	*21,532*	*0*	*2,665*	*0*

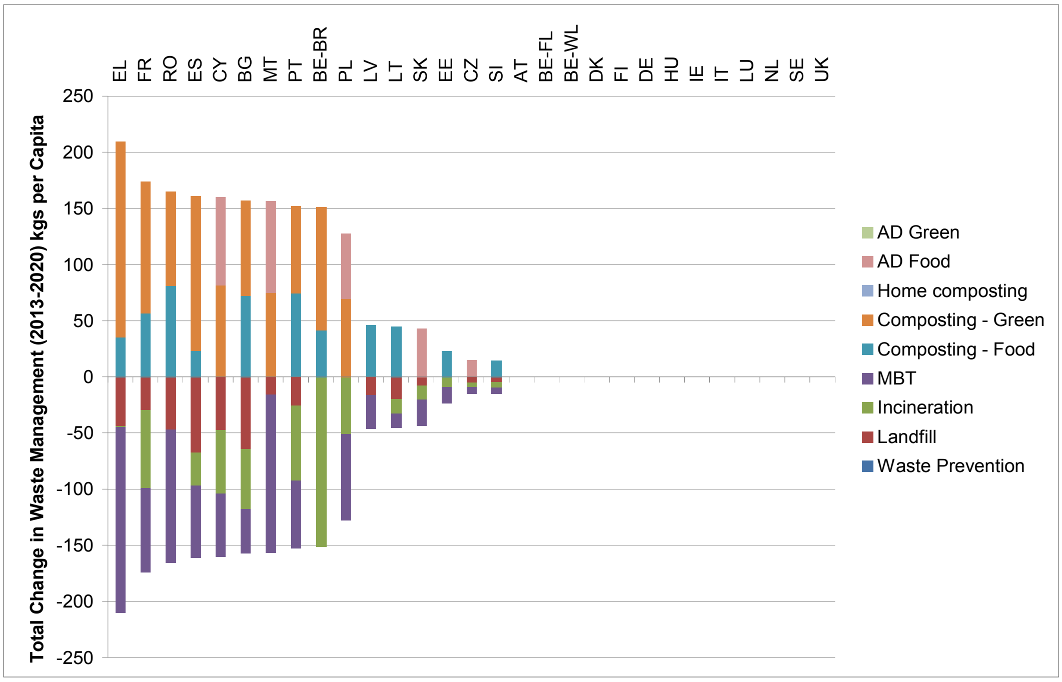
The table indicates that approximately 7.9 million tonnes of additional biowaste treatment is required, across the EU, by 2020 to meet the targets in Scenario 3. Since the baseline is assumed to meet Landfill Directive targets, this total capacity also represents the minimal additional biowaste diversion beyond Landfill Directive targets. This net reduction beyond Landfill Directive targets is shown for each member state in Figure 10-1. This chart also applies to Scenario 3a since targets under the two scenarios are the same, it is simply the treatment systems which may be different.

Figure 10-1: Reduction in biowaste to landfill beyond Landfill Directive targets for Scenario 3 and 3a (total 2013-2020)



The pattern of waste movements by country is shown by Figure 10-2. As can be seen there is a much greater relative change in garden waste than for Scenario 2. Furthermore, some countries, such as Austria, Denmark, Italy and the UK, are already meeting the targets, so no further separate collection of biowaste is required. This is clearly indicated by the chart.

Figure 10-2: Scenario 3 - Total Change in Waste Management (2013-2020), kgs per Capita

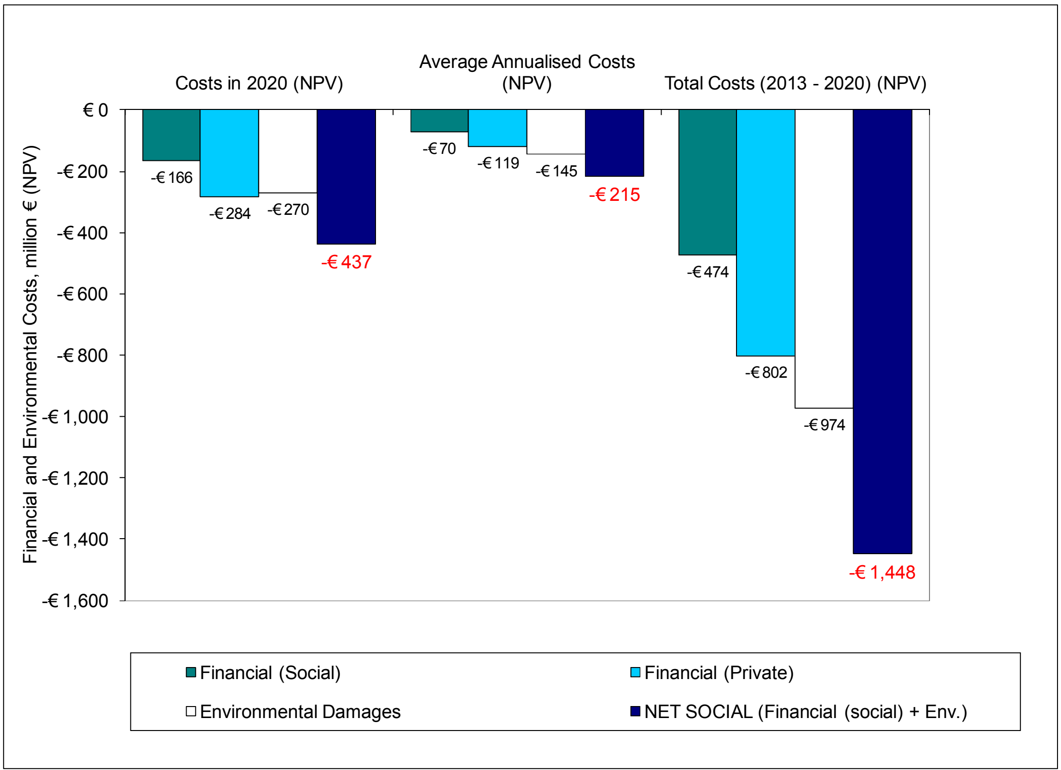


10.2.1 Financial and Environmental Costs of Scenario 3

Figure 10-3 presents the total financial and environmental costs of Scenario 3 for all 27 Member States combined, over the period 2013-2020. The graph indicates that Scenario 3 results in a significant net benefit to society, calculated from adding together the environmental damages and the financial cost under the social metric. **Scenario 3 gives a much lower net social benefit relative to Scenario 2, this being €1,448 million (as compared with €7,088 million for Scenario 2).** The chart also shows the data for the situation in 2020 and the average annualised costs between 2013 and 2020. This shows that the greatest benefits from the policy accrue towards the end of the time period modelled in this study.

As with Scenario 2, these net social benefits relate partly to cost savings and partly to environmental benefits. However, in this case, it is the cost savings which dominate, partly as a result of the assumed absence of any waste prevention effect. As with Scenario 2, the environmental benefits would be higher if the analysis suggested that the preferred option for treating biowaste, from the perspective of society (financial cost under social metric plus environmental cost), was anaerobic digestion in a greater number of countries. On the other hand, the financial savings would be lower so that the net benefit to society (financial cost under social metric plus environmental cost) would be slightly reduced, but with a greater proportion of the benefit being derived from environmental benefits rather than financial savings.

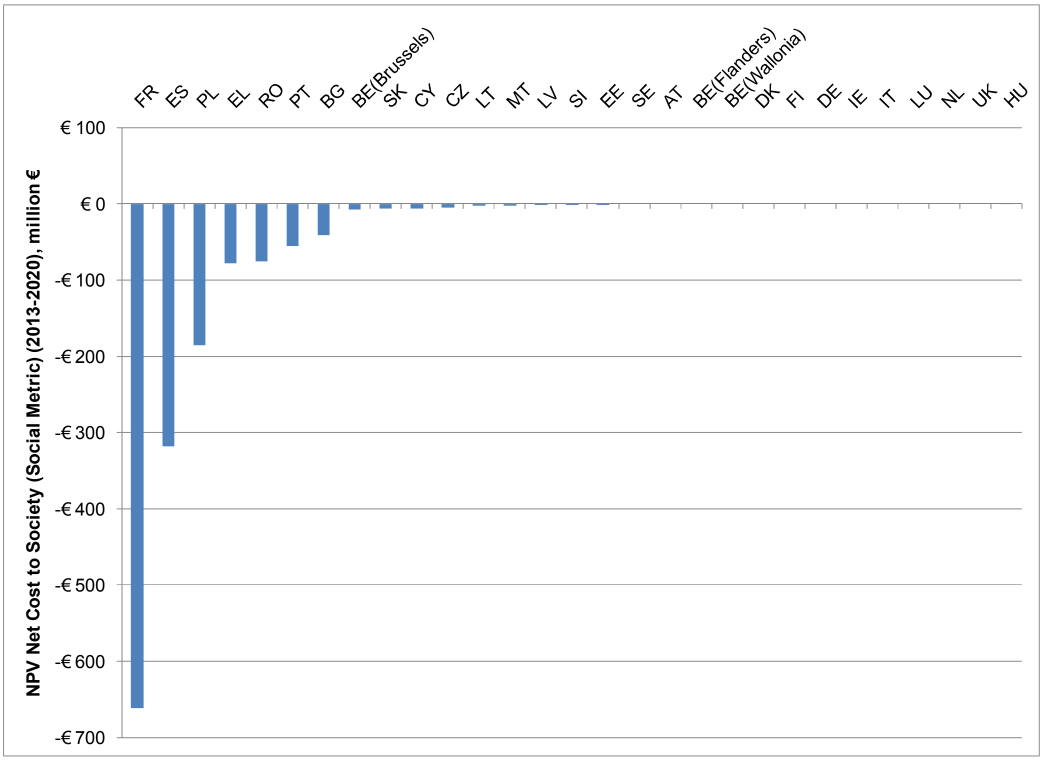
Figure 10-3: Total Financial and Environmental Costs for the EU-27, Scenario 3



Scenario 3 results in a smaller reduction in both the financial and the environmental damage costs. However, the relative proportions of both these costs are very similar to those for Scenario 2. Under this scenario, the quantity of biowaste removed from landfill is less than a quarter of that which is considered to be removed under Scenario 2, and there is no waste prevention impact. As a result, the substantial reduction in environmental damage costs seen under Scenario 2 does not occur to the same degree. There are, however, reductions in the quantities of waste treated by both incineration and MBT facilities. These types of treatment typically result in higher financial costs per tonne of waste treated in comparison to the cost of sending the material to landfill, or composting organic wastes. Thus there is still a reduction in the financial cost associated with Scenario 3 (albeit this is less than half of what was seen under Scenario 2 owing to the smaller changes in tonnage handled by residual waste treatments).

The same rationale for understanding the breakdown of the Net Cost to Society, described for Scenario 2 in Section 9.2, is also relevant here. Hence in this section, just the net costs presented. This is shown, by country, in Figure 10-4 below.

Figure 10-4: Scenario 3 – NPV Net Cost to Society (Social Metric) (2013-2020), million €



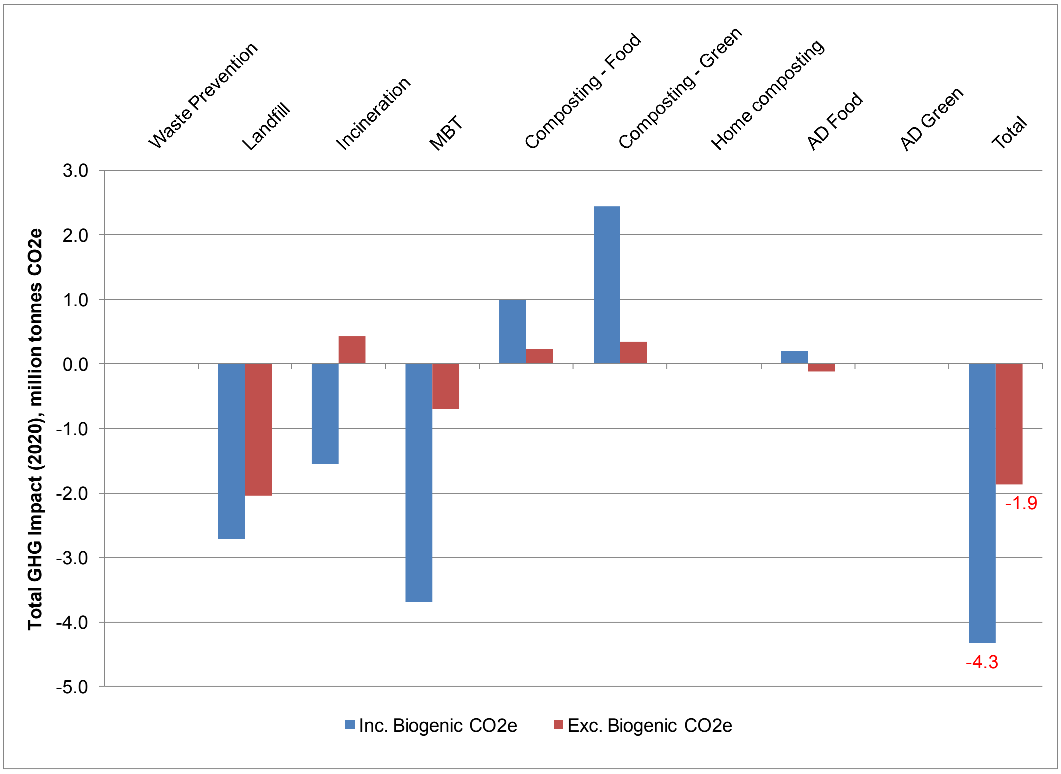
10.2.2 Changes in Greenhouse Gas Emissions, Scenario 3

Figure 10-5 presents the greenhouse gas impacts anticipated to follow from the changes in treatment methods that result from Scenario 3. Results are shown for the year 2020, by which time it is anticipated that the full impact of the policy will be seen. The results on the graph show, for each scenario, the changes in GHG emissions in million tonnes of CO_2 equivalent for the EU-27 in this year. The graphics should be read as showing the changes in emissions from each management route in the scenario. So, for example, where a negative bar is shown against landfill, this does not mean that landfilling implies a net GHG saving. It shows how the change in quantity landfilled translates into changes in GHG emissions.

Scenario 3 shows a far less significant reduction in overall GHG emissions than under Scenario 2. This is true in both the case where emissions of CO_2 eq from biogenic sources are included, and where they are excluded. The less significant change results from the smaller amount being switched from landfill, the absence of any assumed waste prevention effect, and the fact that for many of the countries considered, in net social cost terms, the best case results from deployment of composting as opposed to anaerobic digestion. As highlighted above, in many countries, the situation is finely balanced between whether the best option is compost or a form of anaerobic digestion, with the former costing less, but having greater environmental impact, and the latter costing slightly more, but with lower environmental impact.

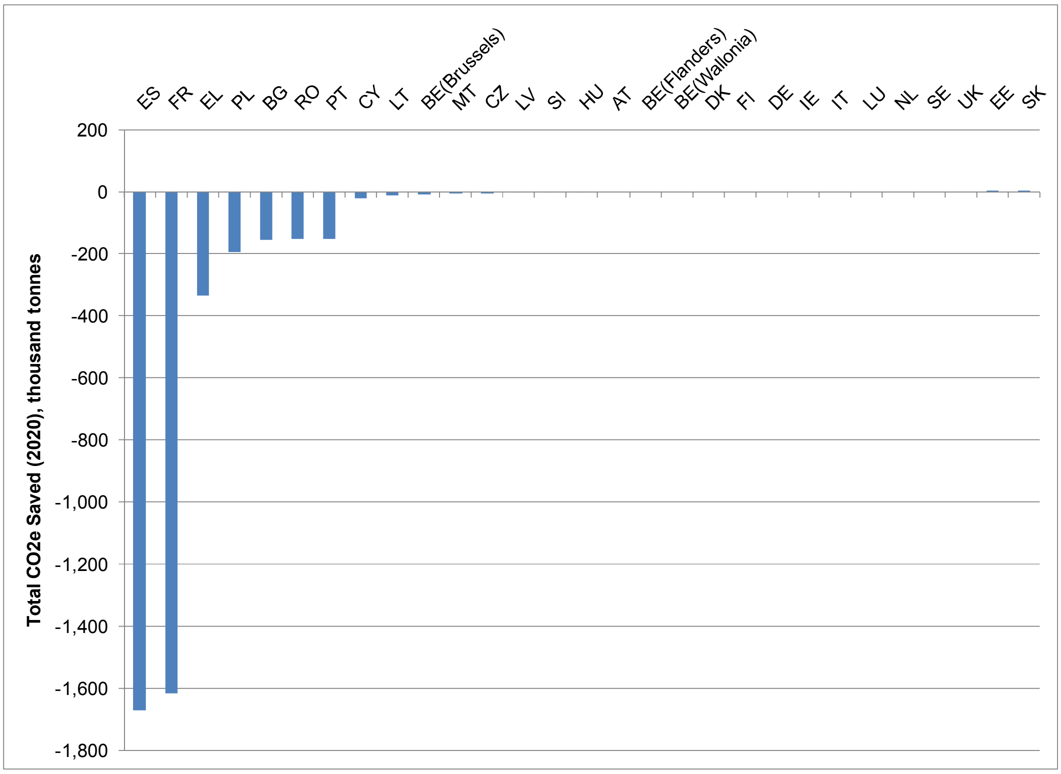
The total benefit is a reduction in 4.3 million tonne CO_2 eq where CO_2 eq from biogenic sources is included in the calculation, and 1.9 million tonnes where it is not.

Figure 10-5: Total Greenhouse Gas Implications of Scenario 3 for the Year 2020 (Including & Excluding Biogenic CO_2 eq Impacts)



The GHG impacts per country are shown in Figure 10-6 below. As with Net Cost to Society, the quantity of GHGs abated is mainly dependent upon how much additional biowaste is to be separately collected, and treated, and the size of the country.

Figure 10-6: Total Greenhouse Gas Implications of Scenario 3 for the Year 2020 (Including Biogenic CO_2 eq Impacts) for each Country



10.3 IMPACT ASSESSMENT – SCENARIO 3a

The following sections identify the changes in costs and benefits, as well as greenhouse gas impacts, of Scenario 3a. Table 10-3 summarises the Scenario under consideration.

Table 10-3: Key Assumptions Underpinning Scenario 3a

	Key Assumptions
Scenario 3a	• No waste prevention • All countries below currently the European target of 36.5% biowaste collection are assumed to reach this target by 2020. This is achieved by increasing garden waste collections up to the maximum of either the baseline situation or 70% of garden arisings, and any further diversion required is assumed to occur through food waste collection. • Countries above the 36.5% target continue as per the baseline collection • **Additional food waste collected separately treated by option with best greenhouse gas emissions outcome for each country**

The best options, in terms of GHG emissions, for each Member State are given in Section 9.3 above.

The mass flows are effectively the same as under Scenario 3. However, there are some differences relative to Scenario 3, and these are shown in Table 10-4.

Table 10-4: Difference in Mass Flows Between Scenarios 3 and 3a

	Waste movements (thousand tonnes)	
	Compost	**Anaerobic Digestion**
2013	0	0
2014	-358	358
2015	-637	637
2016	-913	913
2017	-1,192	1,192
2018	-1,471	1,471
2019	-1,750	1,750
2020	-2,029	2,029

10.3.1 Financial and Environmental Costs of Scenario 3a

Figure 10-7 presents the total financial and environmental costs of Scenario 3a for all 27 Member States combined, over the period 2013-2020. The graph indicates that Scenario 3a results in a significant net benefit to society, calculated from adding together the environmental damages and the financial cost under the social metric.

The differences between Scenario 3 and Scenario 3a are rather similar to the differences between Scenario 2 and Scenario 2a. Relative to Scenario 3:

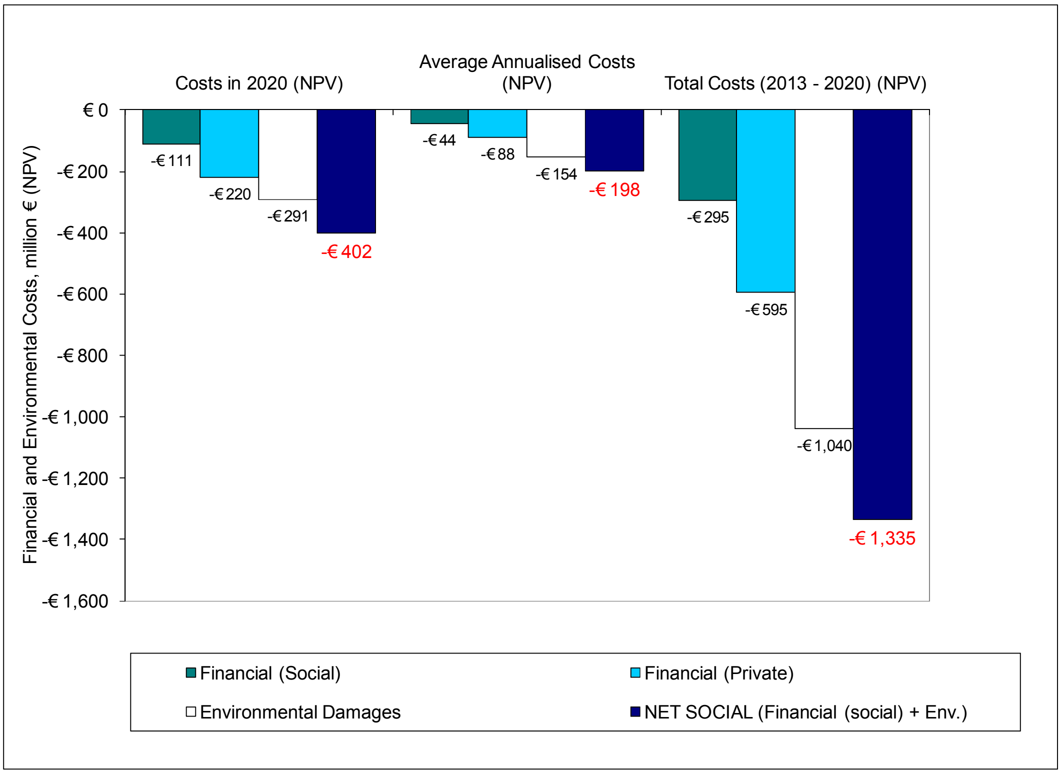
(i) There is a significant increase in environmental benefits;

(ii) However, this is more than offset by the worsening position in financial terms under both social and private metrics;

(iii) The net effect is that net social costs are higher than under Scenario 3 (there is a reduced benefit to society).

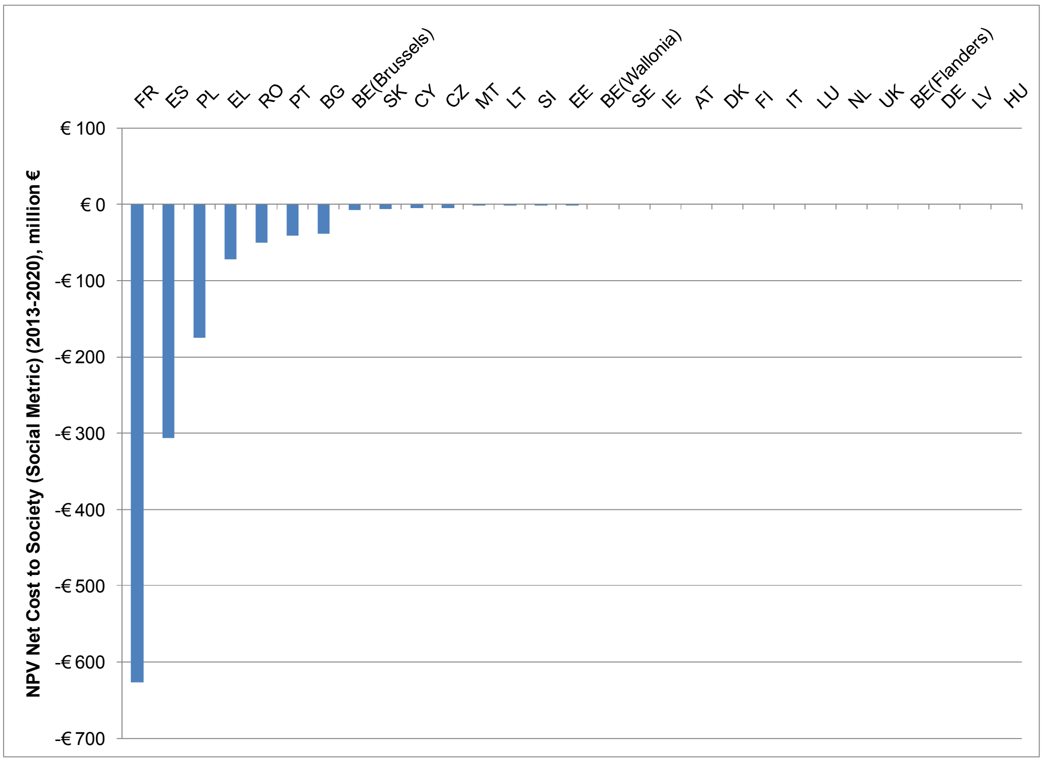
The net benefit position is the worst for all of the scenarios.

Figure 10-7: Total Financial and Environmental Costs for the EU-27, Scenario 3a



The net cost to society for each country is given in Figure 10-8 below. Similar patterns emerge as to those discussed for Scenario 2a in Section 9.3.

Figure 10-8: Scenario 3a – NPV Net Cost to Society (Social Metric) (2013-2020), million €



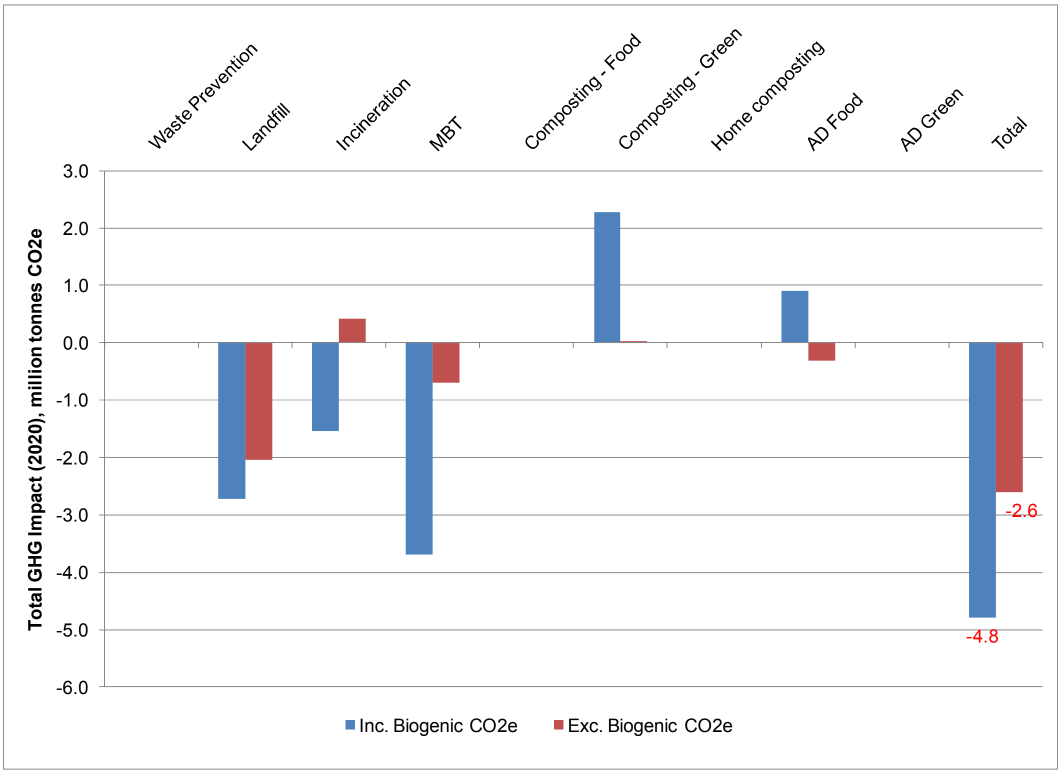
10.3.2 Changes in Greenhouse Gas Emissions, Scenario 3a

Figure 10-9 presents the greenhouse gas impacts anticipated to follow from the changes in treatment methods that result from Scenario 3a. Results are shown for the year 2020, by which time it is anticipated that the full impact of the policy will be seen. The graphics should be read as showing the changes in emissions from each management route in the scenario. So, for example, where a negative bar is shown against landfill, this does not mean that landfilling implies a net GHG saving. It shows how the change in quantity landfilled translates into changes in GHG emissions.

The total benefit is a reduction in 4.8 million tonne CO_2 eq where CO_2 eq from biogenic sources is included in the calculation, and 2.6 million tonnes where it is not (compared with 4.3 million tonnes and 1.9 million tonnes, respectively, under Scenario 3).

As per Scenario 2a, the overall change in GHG abatement is not significant, but there is a greater proportional increase when the biowaste treatments are considered in isolation, especially in terms of the results that are *net* of biogenic carbon.

Figure 10-9: Total Greenhouse Gas Implications of Scenario 3a for the Year 2020 (Including & Excluding Biogenic CO_2 eq Impacts)



11 Summary of all policy scenarios

11.1 Net Cost to Society

The measure most often used in cost benefit analyses and impact assessments is the net cost to society. This is comprised of environmental damage costs and financial costs. In this instance the financial damage costs are those calculated with respect to the social cost metric. It is this figure which is used to determine whether the impact of a policy is positive, or negative, with respect to society. In other words, is the policy worthwhile or not?

To understand where the figure has come from its' main component parts are included in the charts. The financial costs under the private metric are also included, for consideration, as it is these costs that will be seen in the market place. The net present value of both the resultant change in waste management from 2013 to 2020 and the final situation in 2020 is presented. This provides evidence as to the combined effect of the policy over the modelled period (i.e. 2013 to 2020), as well as the further annual benefit that could be achieved through the resultant increase of waste treatment infrastructure required to be in place by 2020, for the targets under each scenario to be met.

Figure 11-1 below shows that under all scenarios there is a *significant* net benefit to society. The greatest benefit is under Scenario 2, at just over € 7 billion - in net present value terms. However, under Scenario 2a, where the best GHG performing biowaste treatment options are considered for each country, the environmental benefits are further increased by 12% (though financial costs are higher).

Figure 11-1: Financial and Environmental Costs of Each Scenario for the EU-27 – Total Cost/Benefit (NPV), million €

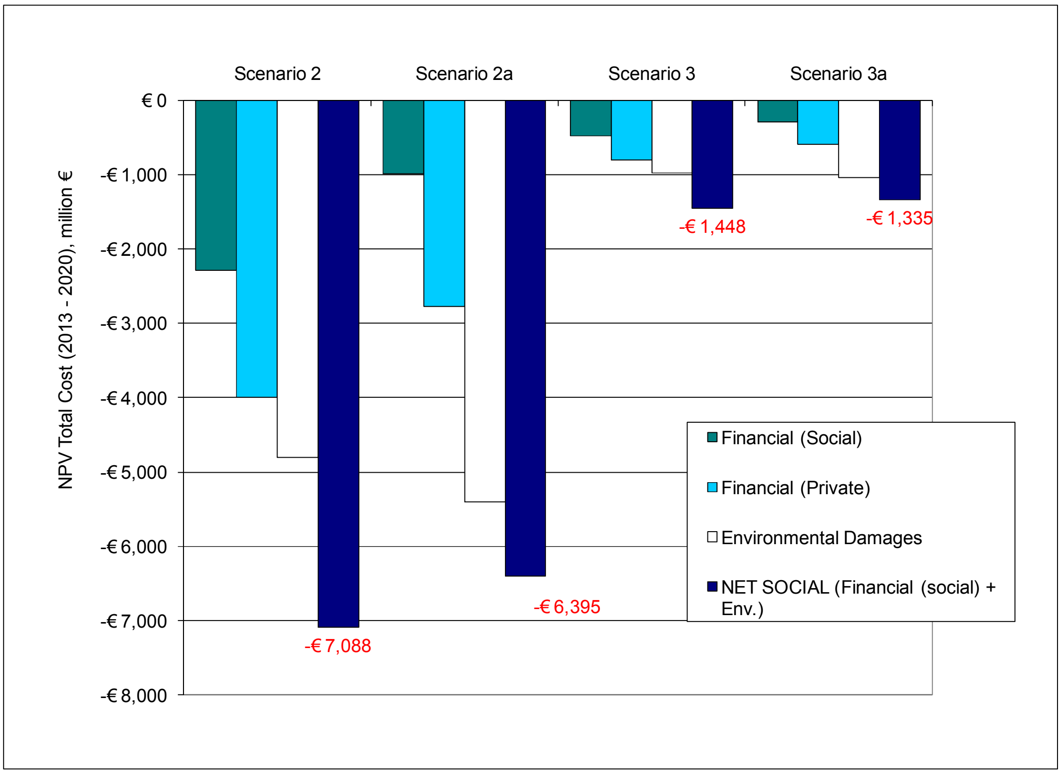
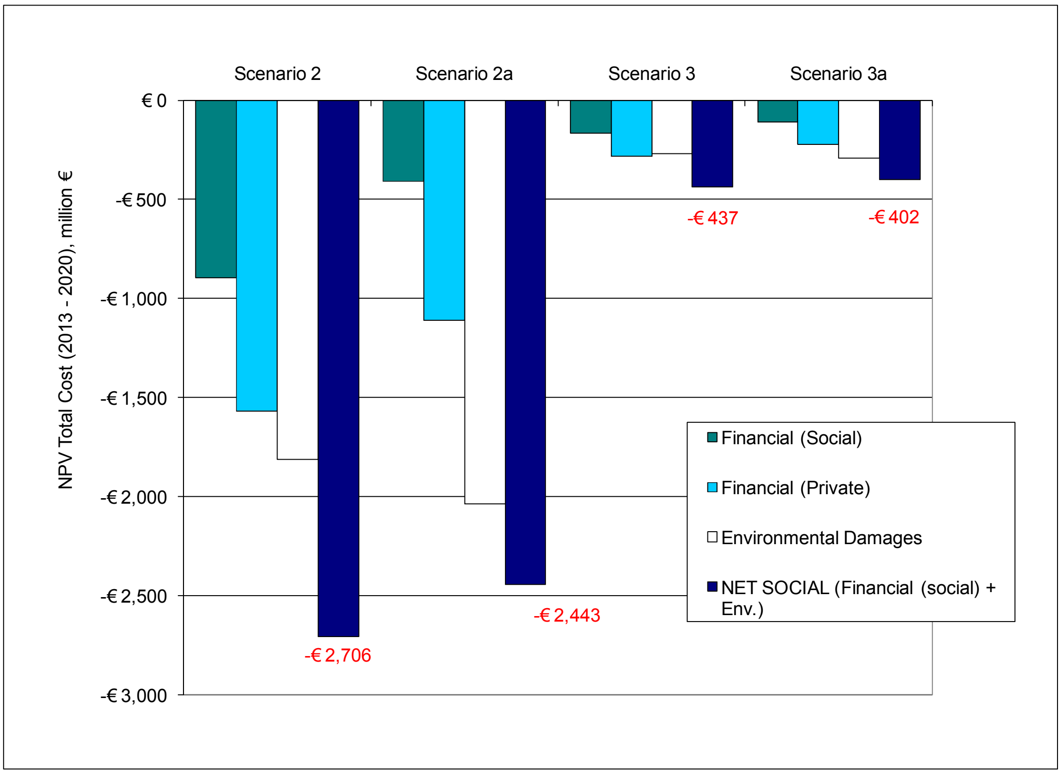


Figure 11-2 below shows that the additional treatment of source separated biowaste develops a significant annual benefit to society from 2020 onwards. This is of great importance since, given that nearly 40% of the total benefit occurs in 2020, the continued benefits, beyond those modelled in this study, will remain significant.

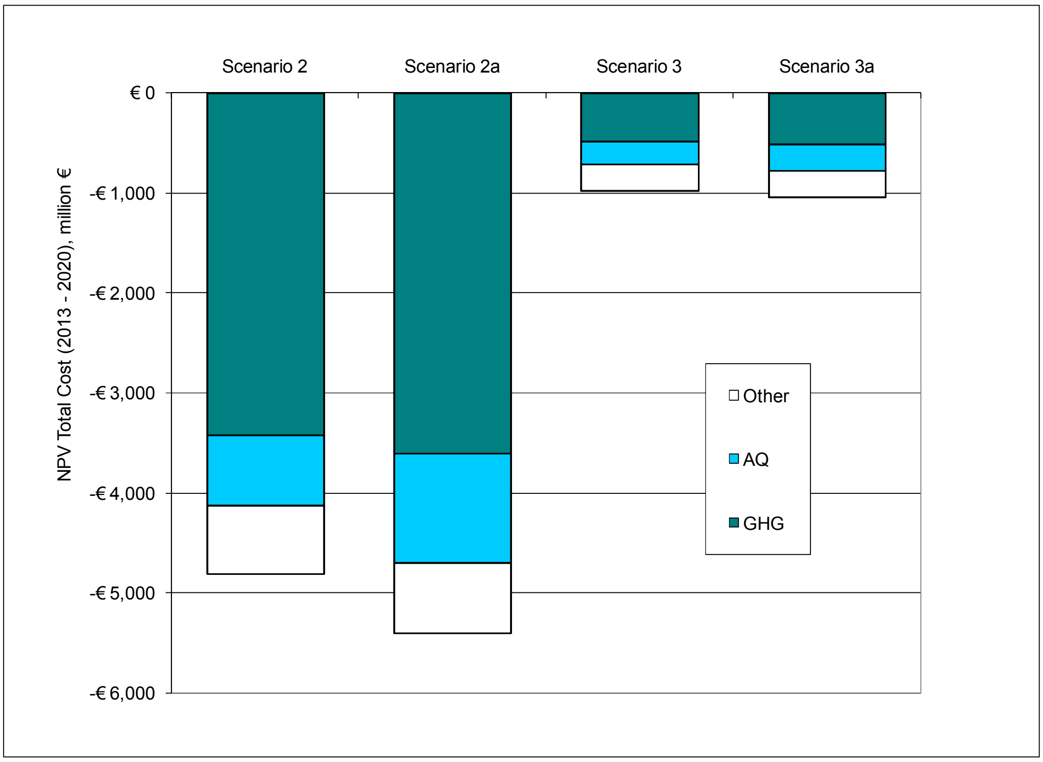
Figure 11-2: Financial and Environmental Costs of Each Scenario for the EU-27 – 2020 Only Cost/Benefit (NPV), million €



11.2 Environmental Damage Costs

In order to understand what factors are the most significant in the determination of the net cost to society, it is important to be aware of the component environmental damage costs. Figure 11-3 clearly shows that the greatest proportion of the benefit to society accrues from the reduction in greenhouse gas emissions. Given that the cost of carbon is likely to increase over time, not decrease, the current snapshot of estimated benefits to society are expected to be understated.

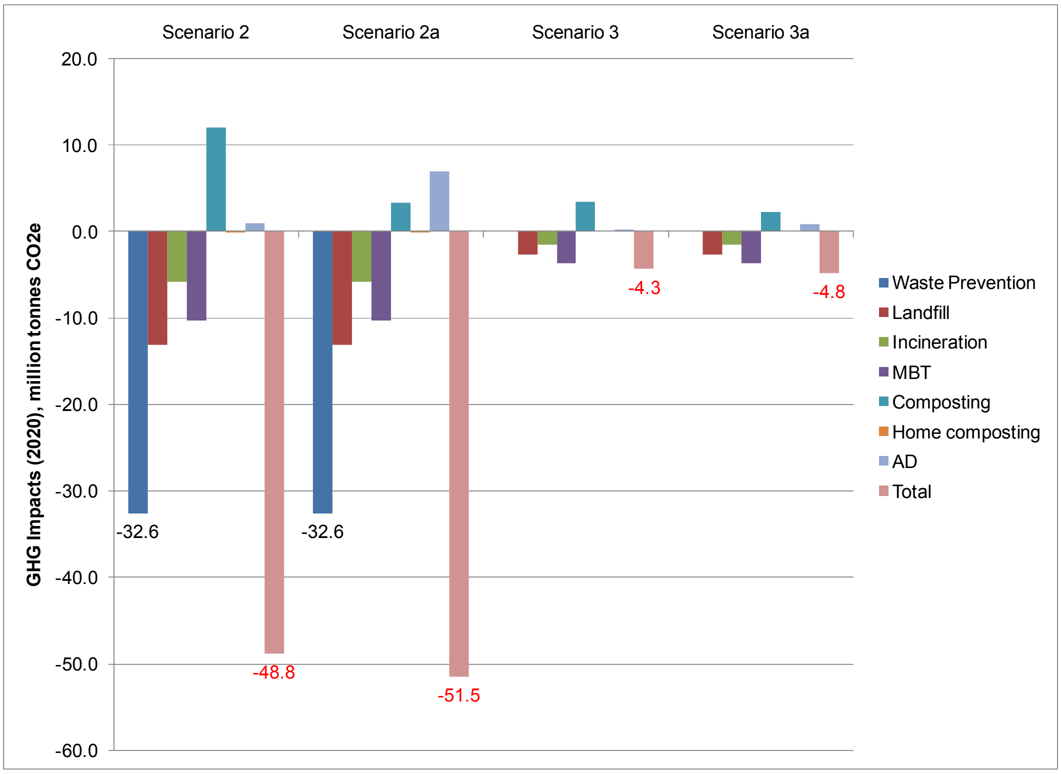
Figure 11-3: Environmental Damage Costs of Each Scenario for the EU-27 – Total Cost/Benefit (NPV), million €



11.3 Total Greenhouse Gas Implications

The chart below shows that under all scenarios, in 2020, the GHG benefits are significant.

Figure 11-4: GHG Implications for all Scenarios in 2020, million tonnes CO_2 eq (Including biogenic CO_2 eq)



The commitments of the Community, as endorsed in the European Council of March 2007, are:

- to achieve at least a 20 % reduction of greenhouse gas emissions by 2020 compared to 1990;
- to achieve a 30 % reduction of greenhouse gas emissions by 2020 compared to 1990 provided that other developed countries commit themselves to comparable emission reductions and economically more advanced developing countries commit themselves to contributing adequately according to their responsibilities and capabilities.

With a 20% target, this means that emissions should be reduced to from 5564 to 4451 million tonnes CO_2 equivalents. With a 30% target, emissions should be reduced to 3895 million tonnes CO_2 equivalents[111].

In the EEA report, two projections are used for 2020:

- In the "with existing measures" (WEM) projections, EU27 GHG emissions in 2020 will be 6% below 1990 levels (5230 million tonnes CO_2 equivalents); this means that under WEM, the 20% target is exceeded by 779 million tonnes CO_2 equivalents;
- In the "with additional measures" (WAM) projections, EU27 GHG emissions in 2020 will be 14% below 1990 levels (4785 million tonnes CO_2 equivalents) this means that under WAM, the 20% target is exceeded by 334 million tonnes CO_2 equivalents;

For the purposes of this study, we shall limit ourselves to the WEM projections.

[111] For the 1990 emissions, we have used Table 2.4 in: EEA, Technical report No 04/2009, *Annual European Community greenhouse gas inventory 1990–2007 and inventory report 2009; Submission to the UNFCCC Secretariat*; Version 27 May 2009.

The following table summarizes, for each policy scenario, the reductions of GHG emissions in 2020 compared to the baseline, expressed in million tonnes of CO_2 equivalents.

Table 11-1: reductions of GHG emissions in 2020 compared to the baseline

	Including biogenic CO_2	Excluding biogenic CO_2
Scenario 2	48.8	40.1
Of which: waste prevention	*32.6*	*32.6*
Scenario 2a	51.5	44.2
Of which: waste prevention	*32.6*	*32.6*
Scenario 3	4.3	1.9
Scenario 3a	4.8	2.6

In other words, if we exclude biogenic emissions, the reductions in GHG emissions under policy scenario 2a could correspond to 6% of the current difference between the 2020 WEM projections and the 2020 targets. About 74% of this contribution would be due to waste prevention effects only. The reduction under policy scenario 2 is only slightly smaller than the reduction under scenario 2a, mainly because the (large) waste prevention effect is the same under both variants.

In the case of policy scenarios 3 and 3a, the possible reduction is more than an order of magnitude smaller (0.25% and 0.3% respectively). It is also noteworthy that the reduction of policy scenario 3a is of the same order of magnitude as the difference between scenario 2 and 2a.

In order to put the contribution of the policy measures analysed in this study in perspective, they need to be analysed in financial terms as well. Whilst there is a lot of uncertainty concerning the marginal abatement costs of greenhouse gasses, most work on the subject acknowledges that they increase steeply within timeframe used in this study. This is certainly the case in some specific sectors such as the transport sector. According to very recent work by the OECD/IEA[112], a reduction of 5 to 16% compared to 1990 emissions (the levels corresponding to the WEM and the WAM projections) would come at a price of 51 to 55 USD[113] per ton CO_2eq. As our results indicate that the financial costs of the policy changes are negative (in other words, that the analysed policy changes all induce financial costs savings), these policy changes can be considered to be “low hanging fruits” in terms of climate policy. However, due to differences in calculations methods used in this study and UNFCCC reporting requirements, these estimates cannot be used directly to assess contribution of biowaste policy to meeting formal GHG reduction targets.

[112] Clapp et al. (2009), National and Sectoral GHG Mitigation Potential: A Comparison Across Models, OECD/IEA.
[113] 2005 USD.

11.4 Limits to the analysis

In the interpretation of the results above, it is important to keep in mind that they are the result of modelling work, and that each model always is a simplification of reality. Although the limitations of our analysis have always clearly been explained in the text, we think it is important to repeat the most important ones:

- There are important **gaps in the data** reported by the Member States. Therefore, there is a lot of uncertainty surrounding even the current state of biowaste generation and treatment.
- There is a lot of **uncertainty concerning the policy intentions of some Member States.**
- The baseline scenario assumes **compliance with the Landfill Directive**. However, as acknowledged by the Commission (see COM(2009) 633 final) several Member States are not currently moving in the right direction.
- There is a lot of **uncertainty regarding the parameters that will influence future costs and benefits** (GDP growth, electricity prices, assumptions regarding financial costs, the choice of damage costs used to assess the pollutants, and other factors specific to different treatment methods)
- Every model involves a trade-off between detail and realism on the one hand and tractability and transparency on the other hand. This requires some aggregation at the geographical and technical level.
- The benefits we have reported have been derived under a central case which assumes the change in collection costs can be constrained to zero. There are good reasons why this assumption would be likely to hold good where collection systems are well designed. However, we note that where collection systems are poorly designed, then additional financial costs may be incurred, and these may eliminate the benefits estimated in the central case.
- The financial savings we have reported are only valid only up until the point where the benefits being delivered still outweigh the potential cost of collection (which we have assumed is zero, but may be more or less than this value).

11.5 Policies

Our analysis has estimated the costs and benefits of reaching some uniform prevention, collection and recycling targets. It has not considered the policies that would be needed at the Member State level to implement these targets.

Possible policies that have already been used in other Member States could include:

- Ordinances for separate collection, requiring local authorities to organise separate collection;
- Targets for recycling and composting / digestion;
- Targets for reducing residual waste;
- Food waste prevention campaigns; and
- Landfill / incineration taxes and bans (which lend support to, rather than drive, such measures)

Subsidiarity implies that while the EU sets the framework, Member States would be free to implement the targets as they see fit in line with subsidiarity.

This could also imply a differentiation in the timeline for the targets. For instance, in order to accommodate the needs for incinerator heavy countries, any policy could incorporate a derogation period for such countries, much as happens with the Landfill Directive targets in cases where Member States have, historically, landfilled very large proportions of municipal waste.

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