

European Commission, DG Environment

**A Study on the Economic
Valuation of Environmental
Externalities from Landfill
Disposal and Incineration of
Waste**

Final Main Report

October 2000

IMPORTANT INTRODUCTORY REMARK

This report is a purely methodological study based on existing information from literature. It is not intended to compare and evaluate various waste management options. **Therefore, this study was neither conceived to compare landfill disposal to incineration nor can any of the results of this study be used to make a generalised statement on which method is to be preferred.**

Executive Summary

Background and Project Scope

In recent years, the European Commission has continuously developed the tool of cost-benefit analysis to better inform decision-makers in the process of settling on new directives and regulations concerning the environment. However, according to the Terms of Reference of this assignment “*most studies in the field of waste have been restricted to an analysis of costs and, at best, a relatively superficial description of benefits*”.

This study aims at reviewing and presenting in an easily comprehensible way the parts of the methodological toolkit of cost-benefit analysis where information is not readily available, i.e. environmental externalities from landfill disposal and incineration. More specifically, the study aims to provide “*an overview of the environmental externalities that need to be taken into account when evaluating different waste management policies and how they can be integrated into cost-benefit analysis.*”

The study was launched by the European Commission in late 1999, and it has been conducted by COWI Consulting Engineers and Planners AS¹.

The study was carried out as a desk-study where the essential inputs were identified through and provided by an essential and thorough literature review. The study was structured according to the Terms of Reference and consequently the following tasks were covered:

- a literature review on existing studies on economic evaluation of externalities;
- an overview of the types of externalities arising from landfill disposal and incineration of waste;
- a description of the impacts on receptors of the externalities;
- a quantification of the main externalities according to typical scenarios for landfill disposal and incineration of waste both in physical and economic terms;
- the establishment of generic data for landfills and incineration plants;
- discussion of the sensitivity of valuation parameters to national preferences;
- proposals for further research.

These tasks have to a large extent determined the outline of the report (c.f. below).

¹ The Terms of Reference for the study are included as Appendix VIII in the Appendix Report.

The study considers the externalities from incineration and landfill disposal of municipal solid waste. Externalities can be defined as *“the costs and benefits which arise when the social or economic activities of one group of people have an impact on another, and when the first group fail to fully account for their impact.”*²

The study has focused on the following externality types:

- external costs related to greenhouse gases causing climate change;
- external costs of conventional air pollutants and some airborne toxic substances causing e.g. health effects;
- external costs of leachate to soil and water;
- external costs of disamenity effects of the facilities, e.g. visual effects, noise, smell and litter; and
- external benefits from energy recovery.

Externalities related to discharge of wastewater to receiving waters have been described but not quantified.

Literature has mainly been collected in the EU and to some extent in the US and other Western countries. National research and analyses have only been included to the extent that they have been readily available and written in English. The exception from this is Danish sources. A full list of literature is presented at the end of the report.

Summary of the Report Content

In order to allow an easier and faster reading of the main content of the study, the Final Report has been divided in two parts; the present Main Report and the Appendix Report. The Main Report can be read as a stand-alone document, but it is recommended that the reader also turns to the Appendix Report as this report forms an important background document giving much more detailed information than the Main Report.

The Main Report includes a description of cost-benefit analysis and the possible use of externalities in cost-benefit analysis (CBA) in Chapter 2. In order to make a sound decision in a policy context it is important to take into account the full range of costs and benefits i.e. both the internal and external costs and benefits. Even if not all information is readily available, the methodology presented in this study can be used to complement the more easily available elements of CBA and inform decision-makers on known elements, uncertainties and gaps of such analyses. It will also serve to make the underlying choices of political decisions more easily understandable and transparent.

A methodological framework for executing the valuation of externalities is presented (Chapter 3). Externalities are damages or benefits, which are not paid for by the polluter or beneficiary under normal market conditions. They can be

² ExternE (1995).

split into fixed (independent of the quantity of waste) or variable (depending on the quantity of waste) costs and benefits. Most waste externalities such as emissions to air, water and soil are variable external costs. Disamenity effects of landfills and incineration plants are mostly fixed external costs. The main variable external benefit in this context is energy recovery from incineration and landfill gas flaring. The produced energy can replace alternative energy production and reduce emissions there.

The report focuses on disposal of municipal solid waste to landfills and incineration plants (definitions and system description in Chapter 4). An overview of the main emissions from incineration (Chapter 5) and landfill disposal (Chapter 6) and their impacts on receptors is given. For incineration, the main conventional air pollutants are NO_x, SO₂, and particulates and the greenhouse gas CO₂. In addition, heavy metals and dioxins³ are emitted to the air from incinerators. A residual product is left after the incineration process, which is disposed of to a landfill. A residual product (bottom ash, fly ash and air pollution control residues) is left after the incineration process, and it is disposed of to landfill. The incineration process also involves a flue gas cleaning process, which may give rise to externalities to wastewater. For landfill disposal, the main emissions are the greenhouse gases CO₂ and CH₄. Moreover, leachate is emitted from the landfill to the surrounding soil and water. Generally, most of the available information is on air emissions and less information exists on the emissions to soil and water. For both disposal options there is in addition disamenity connected to the facility. By recovering heat and/or electricity emissions from other energy sources can be avoided.

Chapters 7 and 8 describe economic valuation of externalities from incineration. There is substantial literature and research on the quantification and valuation of the impacts of conventional **air emissions** and their resulting damage. The dispersion and impact patterns are relatively uniform once pollutants are emitted. Therefore, it is relatively easy to generalise the damage estimates and to apply such estimates widely. It can thus be concluded that valuation results in this field - covering both the external costs and the external benefits - can be considered to be quite comprehensive and fairly robust although they are of course still subject to uncertainties. These uncertainties are reflected in relatively wide ranges of estimates. Other air emissions such as heavy metals and dioxins are, however, quantified relatively rarely.

However, very few attempts have been made to quantify and value **soil and water externalities** from landfills. Pollution pathways of emissions to soil and water are quite site specific and difficult to measure. They depend largely on the quality of the soil, and on the specific location of the landfill with respect to for example groundwater reservoirs and receiving waters. Therefore, calculations on soil and water externalities must be considered as highly uncertain.

³ "Dioxins" refers to the total concentration of dioxins and furans, including PCDDs and PCDFs,

Furthermore, the knowledge of the **long-term effects** from especially contained landfill sites is highly limited today due to the mere fact that such sites have not existed for very long.

Damages may happen several decades after the emission has occurred. This raises the question of **discounting** and intergenerational distribution. How should a damage that occurs today be valued compared to a damage that happens in the future? Discounting is continuously subject of debate and the choice of the discount factor is very important for the result of a CBA or a valuation study.

For **disamenity** externalities, a number of studies were conducted in the US in the 1980s and early 1990s (especially for landfills). In Europe, however, similar studies have not been undertaken on a larger scale. Only two European studies have been identified in the course of this project.

For all of the mentioned externalities, the question of **benefit transfer** is relevant. The willingness to pay or the willingness to accept may differ between countries. In this study, US valuation figures for disamenity impacts of landfills and incinerators have been used as the basis to calculate EU estimates. This is the best possible approach in the absence of available EU figures. It is likely, however, that the preferences differ between the various countries and adjustments need to be made for a more exact analysis.

The main externalities have been quantified in **five calculation examples** to provide illustrations of the results that can be obtained relying on the current state of knowledge (Chapter 9).

The five examples cover both incineration and landfill disposal and reflect different environmental standards and different levels of energy recovery. They cannot be seen as representative for incinerators or landfills all over the European Union as no attempt was made to verify whether the assumptions used correspond to practical conditions. There was furthermore not made any attempt to differentiate between national preferences, which may play a significant role. The examples take account of uncertainties in both emission data and valuation data, which implies that the results are generally presented as a main value with indication of an interval around this value reflecting the uncertainty in data.

In spite of the above caveats, the following conclusions seem to be relatively robust: There is **no easy and straightforward answer as to whether incineration or landfill disposal is preferable** from the point of view of external effects. However, stricter standards of both landfills and incinerators can help to reduce such effects substantially. In particular with respect to incinerators, the alternative energy source replaced by recovered energy from waste plays a very important role which may change incineration from a clearly favourable position to an unfavourable position and vice versa.

There is a slightly higher uncertainty for the evaluation of the relative importance of the various external effects but the following results can be seen as

relatively clear given the current state of knowledge. For incinerators, the most important effects seem to be classical air pollutants whereas disamenity and global warming effects seem to be of a somewhat smaller dimension. For landfills, the major effects seem to be disamenity and global warming.

According to current knowledge, leachates play a minor role as regards the overall externalities. However, there is a high degree of uncertainty linked to this judgement and further research is required to confirm this pattern.

The Appendix Report provides more detailed information to most of the chapters in the Main Report. The Appendix Report presents a detailed outline of the emissions from incineration (Appendix I) and landfill disposal (Appendix II) and their impacts focusing also on the pollution pathways (also referred to as the mapping of externalities). Economic valuation techniques for externalities are presented along with other technical subjects related to valuation such as benefit transfer (Appendix III). This appendix is an important foundation for the understanding of the valuation results presented in this report.

The full literature review of valuation studies on externalities from incineration (Appendix IV) and landfill disposal (Appendix V) is presented. This review forms the basis for the results presented in Chapters 7 and 8. The detailed background information and assumptions for the examples in Chapter 9 are presented (Appendix VI). In addition, generic data on landfills and incineration plants in the EU are presented (Appendix VII). These data provide a rough indication of the number and size of facilities in the EU. These data are not considered as complete, mainly due to the lack of proper data collection within this field.

Conclusions and Recommendations

In general, the methodology to value externalities from air pollution is fairly well developed and can be applied with a relatively high degree of confidence. For disamenity effects and, in particular, emissions to water and soil, there are relatively few attempts of economic valuation and the existing data and methodologies should be considered as highly uncertain.

Despite all the uncertainties in the quantification of externalities from waste disposal, this study has pointed out that preliminary estimates of external costs and benefits can be established. Such estimates cannot be presented as exact values, but they can be used as decision support and as an instrument to explain the trade-offs that are implicitly made in political decisions. Nevertheless, the uncertainties surrounding the estimates should be clearly explained in applied CBA studies and need to be understood if such studies are used to back up decisions.

There is substantial need for further research and substantial efforts should be put into establishing data required for more exact analyses. The largest research gap is on pollution pathways and dose-response functions of pollutants to soil and water. More information is also necessary on the valuation of disamenity

impacts in European countries as well as information on national preferences in the context of possible benefit transfer. The issue of discounting in the context of long-term effects of waste disposal needs to be clarified. There is little generic information about the landfills and incineration plants in the EU in a comparable and reliable format. Without such information, the uncertainties in evaluating the effects of policies over the whole European Union will remain substantial.

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List of Abbreviations

CBA	Cost-Benefit Analysis
LFG	Landfill gas
MSW	Municipal solid waste
RAD	Restricted activity days
TDI	Tolerable daily intake
ToR	Terms of Reference
VOLY	Value of life year lost
VOSL	Value of a statistical life
VSL	Value of a statistical life
WTA	Willingness to accept
WTP	Willingness to pay
YOLL	Years of life lost

Chemical Symbols

As	Arsenic
Cd	Cadmium
CH ₄	Methane
Cl ⁻	Chloride
Co	Cobalt
CO	Carbon monoxide
CO ₂	Carbon dioxide
Cr	Chromium
Cu	Copper
F ⁻	Fluoride
H ₂ S	Hydrogen sulphide
HC	Hydrocarbon
HCl	Hydrogen chloride
HF	Hydrogen fluoride
Hg	Mercury
Mn	Manganese
MTBE	Methyl Tertiary Butyl Ether
NH ₃	Ammonia
NH ₄	Ammonium
Ni	Nickel
NO ₂	Nitrogen dioxide
NO _x	Nitrogen oxides

Pb	Lead
PCBs	Polychlorinated bi-phenyls
PM ₁₀	Particles with a diameter of <10 µm
PO ₄ ⁻	Phosphate
Sb	Antimony
Sn	Tin
SO ₂	Sulphur dioxide
SO ₄ ⁻	Sulphion
Tl	Thallium
TOC	Total organic carbon
V	Vanadium
VC	Vinyl Chloride
VOC	Volatile organic compound

1 Introduction

1.1 Study Background, Purpose and Context

The European Commission has in recent years continuously developed the tool of cost-benefit analysis to better inform decision-makers in the process of settling on new directives and regulations concerning the environment. However, according to the Terms of Reference of this assignment “*most studies in the field of waste have been restricted to an analysis of costs and, at best, a relatively superficial description of benefits*”.

The present study aims to assist the European Commission, DG Environment in improving its state of knowledge on environmental externalities from landfills and incineration. More specifically, the study aims to provide “*an overview of the environmental externalities that need to be taken into account when evaluating different waste management policies and how they can be integrated into cost-benefit analysis.*”

In carrying out the study, this aim has been translated into the operational objective below. This objective has been used as a benchmark against which the outcomes of the various activities have been assessed.

The intermediate objective of the study has been defined as follows:

- To provide an overview of and critically evaluate the existing knowledge regarding externalities from incineration and landfill disposal in order to assess the level of available information and identify important gaps in knowledge. More specifically, to give an **overview** of the types of externalities arising from incineration and landfill disposal of waste and describe the **impact on receptors** from these externalities. To present a critical **literature review** of economic valuation of externalities from incineration and landfill disposal and, finally, to present **examples** quantifying the main externalities from the two waste disposal types.

This objective is seen merely as a further operationalisation of the objective of the Terms of Reference. Its prime purpose is merely to facilitate an operational targeting of the study activities and to strengthen and make explicit the relations between the various activities.

The study is carried out in the period from January 2000 to September 2000.

1.2 Study Delineation

The study considers externalities from disposal of waste to landfills and incineration plants. Hence, it does not cover other types of waste disposal such as recycling and biological treatment. The study serves to provide an overview of the existing knowledge and literature about the externalities rather than to provide a comparative assessment of the externalities from landfill disposal and incineration of waste.

The study focuses on waste that originates from households and household-like waste from commerce and industry. This is commonly referred to as municipal solid waste (MSW). This implies that the study does not consider specific types of waste such as waste classified as e.g. industrial waste, construction and demolition waste, or agricultural waste. Another implication is that the study does not include waste, which is in itself classified as hazardous waste. However, fractions of the MSW may consist of hazardous substances. In this case, hazardous substances are covered by the study.

The study is confined to considering externalities that are caused by normal operation of landfills and incineration plants. Hence, the study does not consider the externalities that are caused by collection, transport and pre-treatment of the waste.

1.3 Outline of the Final Report

According to the Terms of Reference the purpose of the Final Report is to present the results of the study as proposed by the Consultant.

Because of the vast amount of relevant information for the study, it has been decided to divide the Final Report in two parts; the present Main Report and the Appendix Report. In order to fulfil the objectives of the study the report has been divided into the following chapters:

- Chapter 2 describes cost-benefit analyses and the possible use of externalities in cost-benefit analysis.
- Chapter 3 provides a description of a methodological framework for executing the valuation of externalities.
- Chapter 4 describes the waste management system and defines the term municipal solid waste.
- In Chapter 5, a description of the main emissions from incineration and their impacts is given.
- Chapter 6 gives a similar overview of emissions from landfill disposal and their impacts.
- In Chapter 7 economic valuation of externalities from incineration is described. This is done based on a review of existing literature on the subject.

The purpose of this chapter is therefore not to provide new results for the valuation of externalities, but merely to give a presentation of the existing knowledge.

- In Chapter 8, a similar review is made for externalities from landfill disposal.
- Finally, Chapter 9 quantifies the main externalities in five examples with different assumptions regarding technology and energy recovery for the facilities.

In addition, the Appendix Report goes into more detail for several of the subjects in the present report. The outline of the Appendix Report is as follows:

- Appendix I presents a detailed outline of the emissions from incineration and their impacts focusing also on the pollution pathways.
- In Appendix II a similar presentation is given of the emissions from landfill disposal and their impacts focusing also on the pollution pathways.
- Appendix III presents economic valuation techniques for externalities and describes other technical subjects related to valuation such as benefit transfer. This appendix is an important foundation for the understanding of the valuation results presented in this report.
- In Appendix IV the literature review of valuation studies on externalities from incineration is presented. This review forms the basis for the results presented in Chapter 7.
- Appendix V presents the literature review of valuation studies on externalities from landfill disposal, which forms the basis for Chapter 8.
- Appendix VI presents in detail the background information and assumptions for the examples in Chapter 9 and this appendix is seen as an important part in the correct understanding of the results of the examples.
- Finally, Appendix VII gives generic data on landfills and incineration plants in EU. This data is a rough indication of the number and size of facilities in the EU. This data is not considered as complete mainly due to the lack of proper data collection within this field.

2 Introduction to Cost-Benefit Analysis

There is an on-going process in the Community, which seeks to ensure a sustainable development. The road towards sustainable development involves better integration of economics into environmental decision-making, in particular through the use of economic techniques for the appraisal of projects and policies. A technique central to this effort is cost-benefit analysis (CBA). In recent years, the Community has further developed the CBA tool within most sectors to better inform decision-makers in their decision on new directives and regulations.

Below, the basic elements of CBA are outlined. The description provides a rough idea of how economics, through CBA, can assist environmental policy-making. The description aims to illustrate that the principles of environmental economics can contribute to environmental regulations by providing a better foundation for decision-making. Still, it is fairly general, and it does not attempt to provide neither a complete introduction nor an overview of CBA⁴.

2.1 Cost-Benefit Analysis in Decision-Making

Policy and decision-makers are called upon to make decisions on the basis of the full range of advantages and disadvantages of a particular policy. In order to make sound policy decisions, policy-makers need information about the benefits and costs of alternative options for addressing a particular environmental problem. Once the policy-makers have that information, the CBA framework provides a useful tool for the policy-makers as it facilitates overall appraisal and comparison of alternative project options.

Economic analyses in general and CBA in particular can provide policy makers with valuable information for proper decision and policy making. If a forest is spoiled by air emissions or drinking water is contaminated with pollution, these assets start to become scarce. In this case, people will start to reveal their preferences (WTP) for environmental quality by their decisions to locate or in their spending plans. These changes in behaviour can be used to infer what people would be willing to pay to improve or restore the environmental quality at hand

⁴ Readers interested in a more in-dept description of the usefulness of economic valuation in dealing with environmental matters can find useful information in OECD Environment Directorate Environment Policy Committee (April 1995): *Environmental Project and Policy Appraisal: A Manual for Policy Analysts*.

or to protect it from further deterioration. Through economic analyses, it can be analysed at what price people are generally willing to pay for a certain environmental improvement.

In a perfect world with perfect information, prioritising becomes a trivial task of comparing the net result (advantages minus disadvantages) of the different options. In this respect, the CBA is merely a tool, which judges projects according to a simple comparison between the complete range of costs and benefits.

In practice, it is however almost always problematic to obtain the full range of impacts, especially since all impacts ideally need to be fully quantified in monetary values. In many cases all impacts are not fully quantified, but are limited to those areas for which the monetary value gives value-added information (i.e. where an approximate value is better than no value at all). In these cases care should be taken to interpret the quantitative results of a CBA and the results should be weighed against the data material. In any case, all impacts should be mentioned in an analysis irrespective of quantification or not. This means that it is still better to give a description of the impacts than having no valuation and not mentioning the impact at all.

From an economic perspective the environmental goods must be given the same attention by decision-makers in decision-making as all other goods and services in society. If more resources are allocated to protection of the environment, it will reduce the resources that can be spent on production of other goods and services in society. Economists are committed to the principle that economic efficiency should be a fundamental criterion for public investment and policy-making. This implies that the benefits from the use of the scarce resources are maximised net of the costs of using them. This principle is fundamental to the use of CBA as a decision tool. However, economists also acknowledge that economic efficiency should not be the sole criterion in decision-making. Distributional and competitive considerations can be rational justifications for deviation from the principles of economic efficiency as an absolute criterion for maximising the welfare of society. Also, there is often a lack of knowledge and information about the environmental effects as well as general uncertainty about prices and valuation estimates. All of this can make the result of CBA tentative.

A very important aspect of CBA is that ideally it requires economic values as far as possible for all goods, services and environmental effects associated with a given project or policy. Where a market for a good or service exists, the value can be measured simply by its price (adjusted for any distortions such as e.g. transfer payments). However, for many types of environmental "goods" (e.g. clean air) there are simply no markets and no observable price, so economists have to resort to other methods to value the "goods". This valuation is often complicated and associated with great uncertainty.

However, theoretically the value of environmental quality can be inferred from what people would be willing to pay to improve or restore it, using various valuation techniques that measure people's preferences. Thus, the value of an

environmental good can be derived by measuring people's willingness to pay (WTP) for the good, that is how much they would forego in income to obtain an increase in environmental quality. Alternatively, in cases where environmental quality is under threat, the minimum amount people are willing to accept (WTA) in compensation for the deterioration in environmental quality is a measure of the value of that environmental good or service. Whether one uses WTP or WTA ought to be based on whether or not the rights to the environmental quality are in the hands of those being questioned (the citizens), or those who are seeking to worsen environmental quality. In the former case, one ought to use WTA, in the latter WTP. Valuation of environmental goods and of all effects that do not have an observable price in the market is in any case essential for the execution of a CBA.

When the effects of a given policy or project is well known, and all goods and services, including environmental goods, are valued in a common monetary unit, it becomes possible to compare the advantages and disadvantages of the project or policy. For example, the benefits of improving environmental quality can be compared to the costs of improvement. This is the core of CBA.

In practice, the steps to undertake in a CBA for a project or a policy are identification of impacts, physical quantification, valuation of impacts, calculation of net present value, and, finally, ideally a sensitivity analysis.

It should be noted that in reality this ideal approach can usually not be followed completely. Often there is lack of information. Some impacts can be quantified reasonably well while others can be described in their order of magnitude or potential best. In these cases, it is particularly important to undertake sensitivity and/or dominance analyses in order to show which factors and assumptions influence overall CBA results the most. Further, especially in cases where uncertainty cannot be quantified simply due to lack of knowledge, the quantitative uncertainty analyses should be complemented with more qualitative considerations adding value to the overall CBA results.

2.2 Cost-Benefit Analysis in Waste Management

The regulators of the waste management sector should pay close attention to CBA. It is generally recommended as a tool for policy and decision-makers, but there is also a legal imperative to carry out CBA within the Community decision-making. However, due to uncertainty in the environmental impact assessment and lacking monetary valuation of external effects, CBA is not always carried out in the process of policy-making. This has particularly been the case in the waste management sector, although lately the Community has carried out studies to evaluate the economic consequences of the latest waste incineration directive, etc.

Unless the real costs and benefits of waste management policies, including their impact on the environment, are accounted for through the use of systematic economic analyses, there is a risk that poor policies will be adopted and good policies will be rejected.

This report provides useful information for the policy makers about the values of environmental externalities from landfill and incineration.

2.3 Cost-Benefit Analysis and Valuation of Externalities

The collection and disposal of waste degrades environmental quality and imposes external costs (as well as private costs) on society. The external costs take varied forms: local pollution, transboundary pollution, global pollution, noise nuisances, and visual nuisances. The prices of these impacts are not directly observable in the market. However, the monetary values of the impacts need to be known in order to conduct a proper CBA where all real costs and benefit of policies, including their impact on the environment are fully accounted for.

Unfortunately, up till this point in time, knowledge and integrated information about economic values of the impacts from landfill and incineration have been limited. This is essentially why the present study has been undertaken. The aim of the study is to provide an overview of the available information on the economic valuation of all external effects from landfill and incineration of municipal solid waste.

3 Methodology for Quantifying Externalities

The purpose of this chapter is to present a framework for understanding and analysing externalities from landfill disposal and incineration of waste. The framework is theoretical. For practical purposes it is not possible to quantify all externalities from waste disposal completely using the described methodology. Still, an understanding of the methodology is essential to provide for a deeper understanding also of the workings of externalities. The theoretical framework explained in this chapter also underlines many of the analyses undertaken in this study.

3.1 The Concept of Externalities

Welfare economists aim to maximise individual and social welfare through optimal resource allocation. The concept of externalities has been well established in the theory of welfare economics for more than half a century. However, it is only since the 1960s, that environmental externalities have received a lot of attention, both in terms of quantification and actions to internalise⁵ them. Externalities are defined as:

“The costs and benefits which arise when the social or economic activities of one group of people have an impact on another, and when the first group fail to fully account for their impact.” (ExternE 1995).

External costs and benefits are opposed to “traditional” costs and benefits such as operating costs of an incineration plant or income from sale of energy. Such costs and benefits are also referred to as internal or financial costs. The characteristic of such costs is that they are paid for with a price determined by the market, and this price reflects all the true costs of the good or service that it covers.

⁵ An effect is internalised if the loss of welfare is accompanied by a compensation equal to the damage cost by the agent causing the externality.

3.2 The Theoretical Framework

3.2.1 Grouping of Externalities

When quantifying externalities from disposal of waste, they can be grouped by two dimensions. First, whether the externality is a cost or a benefit, and secondly, whether the externality is fixed or variable.

A *fixed externality* should be understood as an externality that does not vary with the amount of waste. Fixed external costs include disamenity costs and these externalities are better expressed in terms of costs per household or per site rather than costs per tonne of waste. *Variable externalities* are more directly related to the amount of waste generated. Such externalities should be quantified as costs per tonne of waste.

Further, externalities should be divided into *external costs* and *external benefits*. An external cost, or negative externality, is any loss of human wellbeing associated with a process, which is not already allowed for in its price; for example air pollution from incineration. An external benefit (or positive externality) is the opposite of an external cost. An example is the recovery of electricity or heat from an incineration plant. The generated electricity replaces electricity that would have been produced by a conventional source, e.g. a power plant. The avoided external costs of that electricity production constitute the external benefits. Table 3.1 shows the externalities grouped according to the two dimensions.

Table 3.1 *Principal grouping of externalities from landfill disposal and incineration.*

	Variable externality	Fixed externality
Negative externality	Costs of emissions to air, soil and water	Disamenity costs
Positive externality	Avoided costs of displaced pollution e.g. from energy recovery	None ⁶

3.2.2 The Impact Pathway Method

Overall, two key figures are essential for quantifying the variable externalities of a given pollutant:

⁶ The filling of holes by disposing MSW at landfills may be considered a fixed, positive externality.

- Emission factors (measured in kg per tonne of waste)
- Unit costs (measured in EURO per kg emission)

In principle, these factors and knowledge about the quantity of waste can form the basis for quantification of the variable external costs that a tonne of waste will account for.

In addition, the fixed external costs need to be added to obtain the total external costs that the disposal of waste implies.

When quantifying the variable external costs from waste disposal in more detail, the total effects can methodologically be split into a chain of causalities, where each link in the chain is determined independently of the others. This is called the *impact pathway methodology*⁷. The impact pathway methodology traces the passage of a pollutant from the place where it is emitted to the final impact on the receptors that are affected by it.

As illustrated in Figure 3.1 the impact chain can be defined to start with the waste entering the landfill or the incineration plant. Thereby emissions are created which will affect the quality of air, soil, or water by increasing the concentration of the substances in the media. Depending on the type of emission and the location of the facility, a group of receptors (human beings, buildings, animals, etc.) is exposed to the substances in a certain dose depending e.g. on how long time the exposure lasts. This dose will give a negative effect in terms of for example health impacts, and, finally, these impacts will give rise to costs to society.

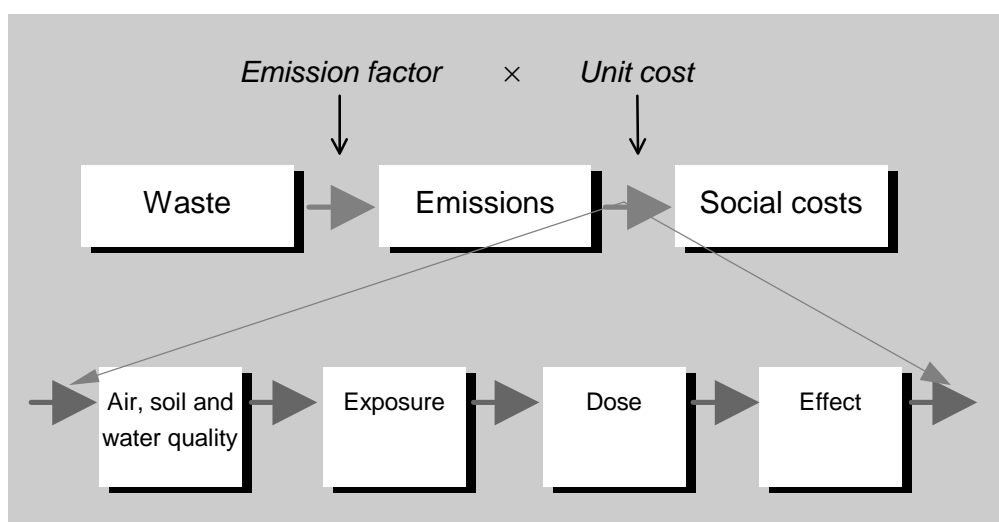


Figure 3.1 Links in the impact pathway from waste disposal to cost.

The impact pathway method represents a bottom-up approach as opposed to a top-down approach, where highly aggregated data are used (for example national emission and impact data) to estimate the costs of the damage of a par-

⁷ Also called the *dose-response technique* or referred to as a *bottom up approach*.

ticular pollutant. The bottom-up approach does therefore in principle⁸ – and contrary to the top-down approach - allow for variation in impacts due to facility, location, or time.

The bottom-up approaches have become much more used in recent years especially within valuing externalities of air pollution due to the increased knowledge of epidemiological data quantifying impacts.

3.2.3 Operationalising the Impact Pathway Method

The impact pathway method as illustrated above is quite complex to quantify in detail. In principle the exposure of each receptor to each of the emission types should be mapped and quantified.

A more pragmatic approach for operationalising the method is necessary, and several assumptions have to be made. In practice, population exposure is for instance used instead of exposure of individuals. On this basis, e.g. the health effects can be calculated based on epidemiological dose-response relations depending on the concentration of the substance. In the same way, the other links in the chain can be operationalised.

The left part of Figure 3.2 shows the operationalised chain. The right part of the figure shows how four factors can be used to calculate the costs of an additional tonne of waste entering either the incineration plant or the landfill. The cost is found by multiplying the four factors: the emission factor, the exposure factor, the exposure-response factor, and the monetary valuation of the damage (the response). Sometimes the exposure factor and the exposure-response factor are gathered in one factor, called the dose-response factor. Further, when a summation of the last parts of the chain is made this is often referred to as a damage function.

The operationalisation of the impact pathway method includes a number of assumptions. For instance, linearity between exposure and damage is often assumed, meaning that e.g. a doubling of the exposure will cause a doubling of the damages. This is normally not the case in reality as a threshold value may exist under which the exposure have no effect. Therefore, the impact pathway methodology is considered more appropriate for situations where only marginal changes are described, as the assumption of linearity does not affect the marginal costs.

⁸ The term in principle is used because the methodology depends on the availability of data from the given site or data for similar sites that can be used as substitute.

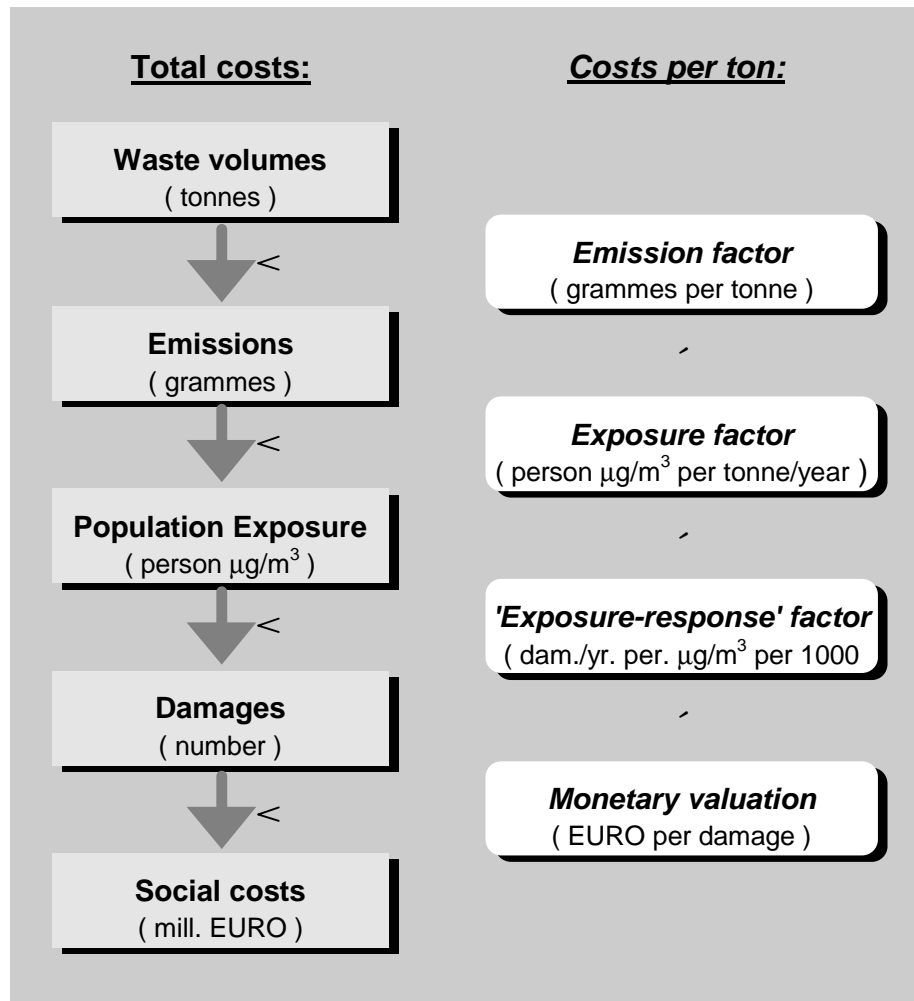


Figure 3.2 Operational impact pathway (for quantifying variable externalities).

3.3 The Study Approach

Quantification of the links in the impact chain can be considered as the ideal way to quantify variable externalities from both incineration and landfill disposal of waste. However, the drawback of this method is the need for very detailed and large quantity of data to build reliable information. This kind of research has been carried out with respect to air emissions, and the foundation for using the impact pathway methodology within this area of externalities is therefore generally accepted.

As for emissions to soil and water, such methods have not been applied. This is mainly due to the lack of knowledge of the transportation and exposure from the pollutants and the damage caused by this exposure, but also due to the site-specific characteristics of the emissions and the impacts.

The presentation of externality costs is based on costs obtained from the literature. The study does not aim at constructing new estimates for the valuation of

externalities, but an outline and comparison is given of the values presented in the literature. In the literature, other types of valuation methods than the impact pathway method have been applied. Furthermore, this method is not relevant for fixed externalities, i.e. disamenity effects. Therefore, other types of valuation results are also presented in Chapters 7 and 8.

4 MSW and the Waste Management System

This chapter introduces the waste management system as it relates to incineration and landfill.

4.1 Definition of Municipal Solid Waste

Focus on municipal solid waste

Waste can be classified in a number of ways such as by material fraction or waste stream (organic, glass, paper), characteristics (combustible, recyclable, hazardous), and source (household, industrial, agricultural etc.). This report focuses on municipal solid waste (MSW), which is waste originating from households as well as household-like waste from commerce and industry.

Waste quantity

Total environmental impacts associated with incineration and landfill are for the most part proportional to the quantity of waste. Total waste generation in the EU and EFTA countries increased by nearly 10% from 1990 to 1995 (EEA, 1998). Based on analyses of current trends, total MSW generation in EEA member countries is thought to continue increasing in the period up to 2010. According to EEA (1998), the proportion of MSW in the EU that is transported directly to landfill (67% in 1995) is unchanged since the mid-1980s, despite efforts to increase recycling levels, incineration, and waste minimisation, because the absolute amount of MSW generated is increasing.

Table 4.1 *Estimated MSW quantities in 1997¹ (total in tonnes and in amount per capita) for European countries.*

Country	Total amounts generated (1000 tonnes)		Amounts per capita (kg/capita)	
	MSW	Of which household waste	MSW	Of which household waste
Austria	4,100	2,775	510	340
Belgium	4,852	-	480	-
Denmark	2,951	2,776	560	530
Finland	2,100	870	410	170
France	28,800	20,800	480	350
Germany	36,976	35,402	460	440
Greece	3,900	-	370	-
Ireland	2,032	1,325	560	370
Italy	26,605	-	460	-
Luxembourg	193	100	460	250
Netherlands	8,716	7,471	560	480
Portugal	3,800	-	380	-
Spain	15,307	-	390	-
Sweden	3,200	-	360	-
UK	28,000	26,000	480	440
Czech Rep.	3,200	2,600	310	250
Hungary	5,000	3,350	500	330
Poland	12,183	8,169	320	210
Slovak Rep.	1,800	1,100	340	200

Source: OECD (1999).

¹Or latest available year.

Waste composition

Environmental impacts of incineration and disposal are influenced by waste composition. A given quantity of MSW is composed of different waste streams or fractions that have varying environmental impacts depending on their inherent hazardous characteristics. For example, the separation or recycling of materials with high contents of heavy metals or organic contaminants decreases the concentration of these pollutants in the remaining MSW stream, thereby reducing the emissions from incineration plants and landfills.

Average MSW composition (by materials and weight) in European countries is shown in Table 4.2. The percentages shown are approximate and can vary with location e.g. between households, regions, and countries, as well as with time e.g. seasonal and yearly variations. Comparisons between countries should be made with care, since significantly different definitions of MSW may be used and different ways of collecting and compiling the information. For instance, it is often difficult to discern whether the MSW composition includes quantities that are recycled or represents MSW sent to incineration and/or landfill.

Table 4.2 *Estimated MSW composition in 1997² (% of total by type of material) for European countries.*

Country	Paper and paperboard	Food and garden waste, etc.	Plastics	Glass	Metals	Textiles and other
Austria	27	27	18	8	7	13
Belgium	16	37	7	7	4	29
Denmark	20	47	5	4	2	24
Finland	26	32	-	6	3	35
France	25	29	11	13	4	18
Germany	41	23	3	22	8	3
Greece	20	47	9	5	5	16
Ireland	33	29	9	6	3	20
Italy ¹	27	32	8	8	4	23
Luxembourg	19	44	8	7	3	20
Netherlands	27	39	5	6	2	20
Portugal	23	35	12	5	3	23
Spain	21	44	11	7	4	13
Sweden	44	30	7	8	2	9
UK	37	19	10	9	7	18
Czech Rep.	8	18	4	4	2	64
Hungary	19	32	5	3	4	36
Poland	10	38	10	12	8	23
Slovak Rep.	15	28	10	6	9	32

Source: OECD (1999).

¹ From 1991 (Source: EC, 1996d)

² Or latest available year.

Focus on incineration and landfill

4.2 The Waste Management System

The handling and disposal of MSW from its generation at source (by the waste producer) to incineration or final disposal in a landfill, can involve a number of intermediary stages such as waste prevention at source, collection, transport, sorting, recycling and biological treatment (see Figure 4.1). This report focuses exclusively on the environmental externalities associated with incineration and landfill disposal once the waste has been delivered to the incineration plant or landfill site.

Integrated waste management

Although the intermediary stages shown in Figure 4.1 are not explicitly taken into account in this report, these stages have an important influence on the environmental impacts related to incineration and landfill of waste. In particular, waste prevention reduces the total amount of waste generated, and recovery or reuse at source (e.g. home composting of organic waste) decreases the amount of waste that would otherwise enter the waste management system. In addition, the waste collection system results in externalities related to i.a. atmospheric emissions from transport.

The environmental impacts associated with incineration and landfilling are also affected by the recovery, reuse and recycling of waste streams, that affect the quantity and composition of waste. For instance, the separate collection and recycling of waste containing high concentrations of heavy metals can significantly reduce the environmental impact of incineration/landfill. Heavy metals are not destroyed during incineration and are either emitted from the incineration plant via the smokestack, in the wastewater, or in residual waste. A reduction in the heavy metal content of waste incinerated therefore results in lower emissions and lower environmental impact.

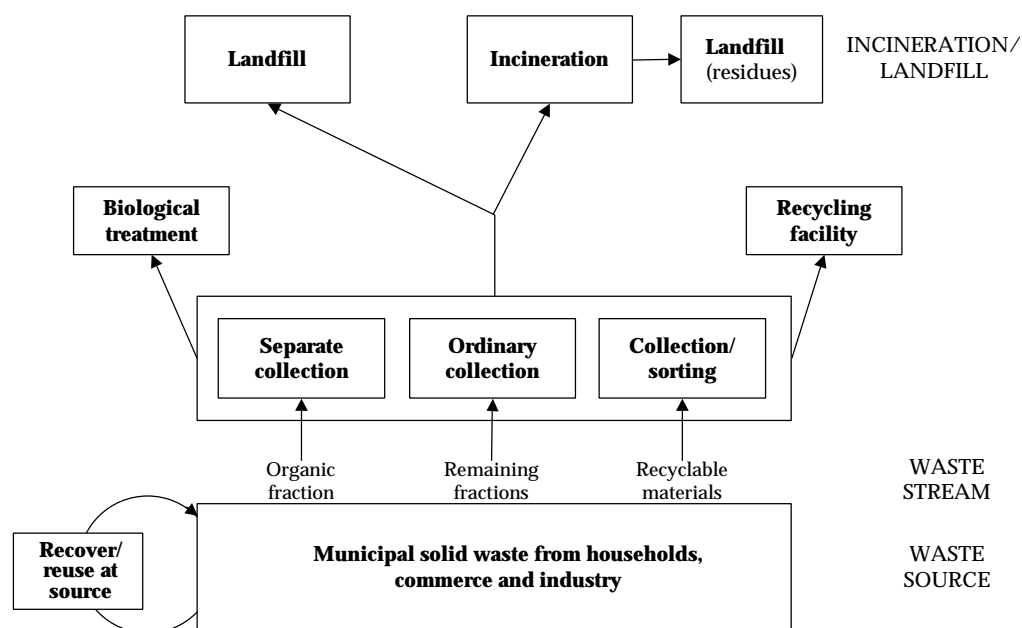


Figure 4.1 Waste management system for MSW from waste source to final disposal.

4.3 Receptors and Damages

Receptors

The impact pathway method involves quantifying the exposure of all receptors to each of the emissions identified. In this study, receptors are defined as organisms in the environment, including humans, fauna, flora, and buildings, that are adversely affected by emissions resulting from incineration or landfill of MSW. Although in principle, all receptors should be identified, in practice the focus is on humans as receptors and the detrimental consequences to human health from direct emissions as well as indirect effects. The reason for this is that the WTP of the population for these impacts usually dominates the WTP for environmental impacts without a link to health effects.

Damages

The damages resulting from various effects on receptors are in this report grouped into the following categories:

- human health effects - mortality
- human health effects - morbidity

- lower agricultural yield
- forest die-back
- damage to buildings
- climate change
- effects on the ecosystem.

The damage categories correspond to the ones in ExternE.

Timing of the damage

Acute effects occur soon after short-term exposure to a pollutant, on the time-scale of hours, days and weeks, and are generally severe (mortality). Chronic effects typically occur as a result of repeated or long-term exposure, although they can also arise following a single exposure, and can lead to mortality or morbidity. The time from the exposure to the effects (latency period) can be relatively long, particularly if the exposure concentration is very low.

Although a pollutant may not have adverse effects in acute toxicity tests, sub-lethal effects (morbidity) may exist. The only way to study chronic sublethal toxicity in the laboratory is by using longer-term exposures over an entire reproductive cycle (Rand and Petrocelli, 1985).

5 Impacts of Incineration

This chapter summarises the emissions from incineration and their impacts on different receptors. A more in-depth analysis is presented in Appendix I of the Appendix Report.

5.1 Inputs and Outputs of the Incineration Plant

Definition

In this report, the term incineration refers to the aerobic thermal treatment of MSW with or without the recovery of energy, and includes the disposal of residual by-products resulting from incineration. Although the specific sequence of unit processes differs between incineration plants⁹, the overall inputs and outputs are similar (see Figure 5.1).

Inputs

MSW and additional resources are inputs to the operation of incineration plants. The additional resources consist both of renewable and non-renewable resources such as auxiliary materials, water, fossil fuels, and land.

Auxiliary materials are used in the flue gas cleaning processes and can include calcium carbonate (CaCO_3) to remove hydrogen chloride and hydrogen fluoride (HCl and HF), and sodium hydroxide (NaOH) to neutralise sulphur dioxide (SO_2). Removal of nitrous oxides (NO_x) and dioxins¹⁰ occurs by injecting activated carbon into the flue gas stream. Additional substances e.g. flocculating agents are used to clean the wastewater produced during the flue gas cleaning process.

⁹ A mass burn incineration plant (grate or fluid bed) is assumed, since such a plant is typically used in the EU and Eastern Europe. Other thermal treatment processes (such as pyrolysis and gasification) are not considered here.

¹⁰ "Dioxins" refers to the total concentration of dioxins and furans, including PCDDs and PCDFs, as defined in Annex 1, EC 98/0289 (COD)

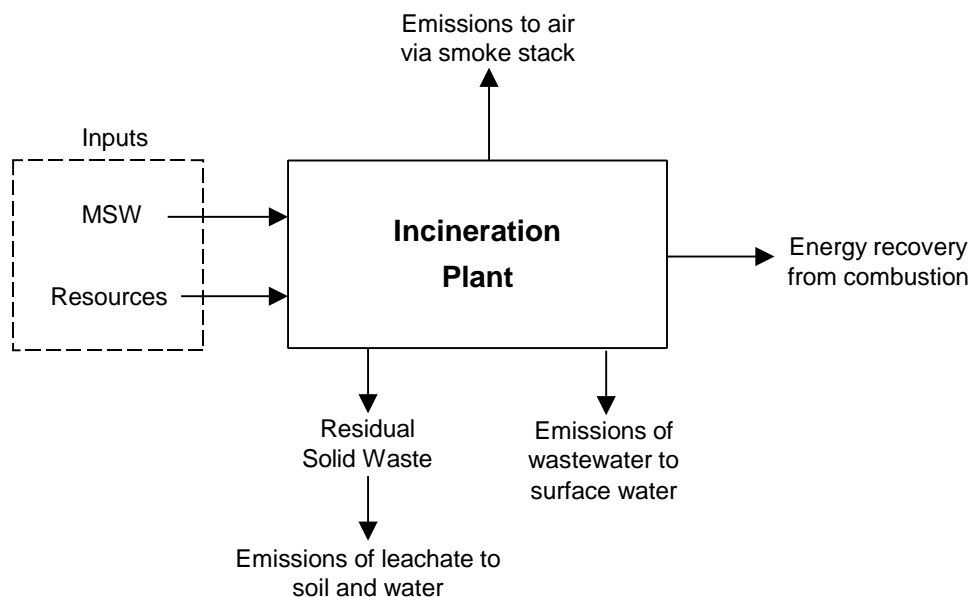


Figure 5.1 Representation of inputs and outputs to an incineration plant that cause environmental impacts.

The quantity of water used is relatively low and can be of secondary quality i.e. not drinking water. The impacts on the environment occur at the local scale and depend on the availability of water.

Fossil fuels such as fuel oil or natural gas are used to start up and shut down incineration plants. The amounts used are relatively low. Electricity, produced from fossil fuels or originating from energy recovered at the plant itself, is additionally consumed during plant operation, particularly by the flue gas cleaning process.

The amount of land required for an incineration plant, which is assumed to be independent of the process technology used, is relatively small compared to the plant capacity.

Outputs

Outputs include emissions to air, water and soil, as well as the energy recovered during combustion. Emissions to air include the flue gas from the incineration process. These emissions can be controlled¹¹ using various treatment process that remove particulates and gases before the remaining flue gas is emitted to the air via the smokestack. Flue gas cleaning processes produce residues that are considered hazardous and need to be treated prior to disposal. The incineration process also generates residual solid waste requiring disposal and/or use e.g. as road construction material. Contaminants in the residual solid waste can be leached and lead to emissions to soil and water.

¹¹ The term "controlled" does not mean that the emissions are completely removed but rather that they are reduced.

The energy produced from incinerating waste can be recovered either in the form of electricity, heat, or both. For example, heat recovered can be used to heat housing. Optimised energy recovery requires a district heating system and/or nearby industry that can use the heat during the summer months when considerable quantities of heat waste would otherwise be wasted. Combined heat and power (CHP) plants that produce electricity and district heating are the most efficient in terms of displaced pollution. However, energy is not recovered from all incinerators, and many, particularly the older incinerators, do not recover energy at all.

5.2 Emissions and Impacts

The impacts of incinerating municipal solid waste relate to emissions of specific contaminants in the flue gas, wastewater and residual solid waste, as well as from energy recovery and the operation of the plant itself.

Emissions to air

Contaminants typically found in the flue gas include particulates, dioxins, heavy metals and their compounds (especially Cd, Tl and Hg), acid gases (SO₂, HCl, HF), nitrogen oxides (NO_x), carbon dioxide (CO) and volatile organic compounds (VOCs). These contaminants are emitted into the atmosphere via the smokestack, although their concentrations can be reduced using flue gas treatment processes. Another emission to air that is influenced by the composition of the waste rather than treatment processes is carbon dioxide.

Once emitted into the air, the contaminants are dispersed in the atmosphere and resulting concentrations depend i.a. on distance from the incineration plant, topography, wind speed and direction, and other climatic conditions, as well as the stability of the substance and its residence time in the atmosphere.

Impacts from air emissions include adverse health effects from particulates, dioxins, heavy metals, VOCs, NO_x, CO and SO₂. Effects on the ecosystem and fauna arise from the same pollutants, especially those that bioaccumulate, such as dioxins, and heavy metals. Lower agricultural yield, forest die-back and damage to buildings can occur from emissions of acid gases and NO_x, with particulates also causing damage to buildings.

Emissions of wastewater

Emissions to water result from the discharge of wastewater from incineration plants with wet flue gas cleaning systems. Wastewater contains many of the same pollutants as those emitted to the atmosphere including suspended solids (particulates), dioxins, and heavy metals. Impacts include human health effects and ecotoxicity.

Residual solid waste

Residual solid waste from incineration plants includes bottom ash and the more toxic residues from flue gas treatment, and is typically disposed of at a landfill site. Emissions to air of dust contaminated with i.a. heavy metals, can arise when handling and landfilling the residues. Air emissions also result from the transport of residual waste from the incineration plant to the disposal site.

Leachate is generated from landfills containing residual solid waste, which can enter the soil and groundwater. Leachate from bottom ash is generally much less contaminated than leachate from flue gas cleaning residues. Leachate from bottom ash initially contains relatively high concentrations of inorganic salts (chloride, sulphate, sodium, potassium and calcium), and concentrations of trace elements (e.g. Co, Mo, Pb, Mn, and Zn) are low. There is a tendency for concentrations of most salts to decrease over time (<30 years) and for the concentration of trace elements to remain low. Initial leachate from flue gas cleaning residues usually contain extremely high concentrations of inorganic salts, especially calcium and chloride, and concentrations of heavy metals (e.g. Pb, Zn, Cd, Cu, Cr, As) are also high. Contaminant concentrations in leachate are thought to decrease substantially with time.

Impacts related to the emission of leachate to the soil include the migration of contaminants to groundwater and/or surface water, where they can affect human health and the ecosystem. At landfill sites with a leachate collection system, emissions to the soil are minimised and treated leachate discharged to the surface water, where potential impacts include human health effects and ecotoxicity.

An overview of the main emissions from incineration to air, water, and soil and the resulting damage is shown in Table 5.1 below. The table gives a general indication of the effect of specific emissions resulting from the incineration of MSW, which does not necessarily correspond to the relative importance of these emissions. Thus, the impacts relate to both normal operating conditions as well as to accidental occurrences. The magnitude of the impacts is considered in relation to specific examples in Chapter 9.

Other impacts related to the incineration of MSW, not shown in Table 5.1, include the displaced impacts resulting from energy recovery and disamenity impacts.

Table 5.1 Overview of knowledge of damage caused by the emissions from incineration, illustrated as dose-response relations.

Emission (dose) \ Damage (response)	Medium	Health effects		Lower agricultural yield	Forest die-back	Damage to buildings	Climate effects	Ecosystem
		Mortality	Morbidity					
Particulates (PM ₁₀)	Air	*	*			*		
NO _x (and O ₃)	Air	*	*	(())	*	*		(())
SO ₂	Air	(*)	(*)	*	*	*		((*)
CO	Air	(*)	(*)				*	
VOCs 1)	Air	(*)						
CO ₂	Air						*	
HCl, HF	Air	((?))	((?))	(())	(())	(())		((?))
Dioxins	Air	(*)	((*)					((*)
Heavy metals	Air	(*)	((*)					((*)
Dioxins	Water	((?))	((?))					((?))
Heavy metals	Water	((?))	((?))					(())
Salts	Water							((?))
Residual solid waste	Soil/ Water	See section Appendix I and II.						

Explanation: * Measurable effect, (*) partly measurable effect, ((*)) non-measurable effect, (()) non-measurable but minor effect, ((?)) non-measurable uncertain effect, blank: no known effect.

1) The effects of VOC related to O₃ and associated damages are included under NO_x.

Energy recovery Net energy recovered from the incineration of waste displaces environmental impacts associated with the production of energy from conventional sources. Displaced impacts include impacts related to atmospheric emissions, as well as the fossil fuels saved, and residual waste avoided from the conventional energy generation process.

Disamenity Disamenity impacts result from the operation of an incineration plant including noise, dust, odours, and visual pollution (particularly the smokestack).

Risk of accidents An accidental or sudden occurrence results in the same types of emissions and impacts as during normal operation, but affects their magnitude. Typically, accidents cause short-term increases in emissions to air, and temporarily reduce or prevent energy recovery.

Impact of incineration The main differences in terms of environmental impacts between incineration plants result from the flue gas treatment process and energy recovery method employed. The flue gas treatment process influences the quantity of atmos-

pheric emissions, wastewater emissions and toxicity of residual solid waste. However, from a mass balance perspective, the total mass of a given pollutant is constant, such that pollutants removed from the flue gases, necessarily end up either in the wastewater and/or in the residual solid waste. This point is especially important when valuing the damage of emissions in economic terms as the valuation of air pollutants is more advanced than the valuation of externalities to wastewater or residual solid waste. Therefore, the emissions of the incinerator are shifted from an economically valued domain to one, which is unvalued.

6 Impacts of Landfill Disposal

This chapter summarises the emissions and impacts on different receptors of landfill disposal.

6.1 Inputs and Outputs of Incineration

Definition

In this report, the term landfill is used to refer to the disposal of MSW to soil, or the site at which such waste is deposited (landfill site). Although the specific design and operation of a landfill varies between sites, the general inputs and outputs related to landfilling are shown in Figure 6.1 below.

Inputs

MSW and additional resources are needed to operate landfill sites. The additional resources consist both of renewable and non-renewable resources such as auxiliary materials, fossil fuels, and land.

At controlled landfill sites where leachate is collected, auxiliary materials are used to treat the leachate prior to recirculation in the landfill, or discharge either to a sewage treatment plant for tertiary treatment or directly to surface water.

During the landfilling process, fossil fuels are consumed by vehicles working at the site, and electricity is required e.g. to operate the weighing station. Once the landfill is closed, energy is required during the active phase for monitoring activities. At modern landfills, energy is also expended to collect and treat leachate, and to collect and use or flare landfill gas.

The amount of land taken up by a landfill is assumed to depend only on the site's waste capacity, i.e. the amount of waste that can be landfilled at the site.

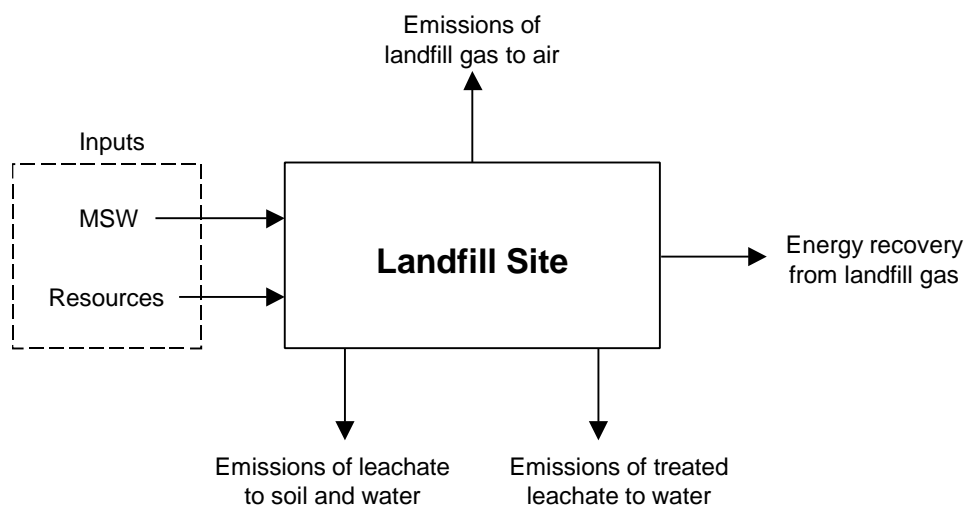


Figure 6.1 Representation of inputs and outputs to a landfill site that cause environmental impacts.

Outputs

Outputs from landfill sites include emissions to air, water and soil, as well as the energy recovered from landfill gas. Landfill gas is emitted to air and leachate generated from the MSW can be emitted to soil and water. At landfill sites with gas collection systems, landfill gas is recovered and used to generate either heat, electricity or both. Where a landfill site has a leachate collection system, emissions to soil and water are reduced, and treated leachate is discharged to surface water.

Knowledge base

Most of the knowledge accumulated about outputs from landfills is relatively limited and based on approximately 30 years of monitoring data from old/uncontrolled landfills receiving all kinds of waste (not just MSW). Relatively little is known about emissions in the long term i.e. more than 30 years, emissions from modern/controlled landfills after site closure, or the effects of changes in waste management policies e.g. separate landfilling of different waste types (Christensen and Kjeldsen, 1995; Kruempelbæk and Ehrig, 1999).

6.2 Emissions and Impacts

The impacts of disposing municipal solid waste to landfill relate to emissions of specific contaminants in landfill gas and leachate, as well as from energy recovery and the operation of the landfill itself.

Landfill gas

Landfill gas is emitted to air and varies in quantity and quality over time. Landfill gas production is thought to be at a maximum shortly after the closure of a landfill and reach insignificant amounts after about 25 to 30 years. The main components of landfill gas are methane (CH₄) and carbon dioxide (CO₂). Trace gases are also present and over 100 different types of VOCs have been identified such as benzene and vinyl chloride.

Landfill gas is either emitted to the atmosphere through the top and/or sides of the landfill and is then dispersed in the atmosphere. Methane and carbon dioxide cause climate effects and VOCs that are toxic and/or carcinogenic may cause health effects. Other emissions to the air include dust that can occur during the operational stage of a landfill, and that may also cause adverse health effects.

Where landfill gas is collected and flared or used to recovery energy, combustible and most trace organic components in landfill gas are destroyed. However, low levels of pollutants not previously present in landfill gas are formed during the combustion process, such as dioxins, which can cause a wide range of health effects.

Leachate

Leachate generated within a landfill is emitted to soil and water. Leachate quantity and especially quality varies over time. The quantity of leachate generated depends mainly on the net precipitation and the type of landfill cover. During the initial phases in the lifetime of a landfill, leachate typically contains very high concentrations of organic carbon, ammonia, chloride, potassium, sodium and hydrogen carbonate, whilst concentrations of heavy metals and specific organic compounds are relatively low. Leachate composition data is only available for certain landfills for a period of about 30 years, however leachate generation may continue for several hundred years, and the long-term potential for remobilization of e.g. heavy metals from a landfill is unknown.

Once released into the soil, processes that affect the transport of contaminants include advection and dispersion, retardation (sorption) and possible degradation of the contaminants to other species. Contaminants can enter the groundwater and/or surface water, where they can affect human health and the ecosystem.

At landfill sites with a leachate collection system, emissions to the soil, groundwater and surface water are minimised. Leachate is treated at the landfill site and/or at the local wastewater treatment plant. Potential impacts including human health effects and ecotoxicity can occur related to the treated leachate that is discharged to surface water, but also the sewage sludge, which can be disposed of to landfill, incinerated or spread onto agricultural land.

The alleged health effect suffered by people living in the vicinity of landfills is a cause of considerable concern. However, most studies relate to hazardous waste dumps or contaminated land, and where studies have been carried out "no causal links have been established and no exposure pathways identified" (Heasman, 1999).

Studies on the health effects of landfills typically do not consider exposure pathways, but take distance from the site as exposure. Epidemiological studies that have been carried out include numerous studies in the US on chemical waste dumps and the EUROHAZCON programme. The latter was initiated to determine the risk of congenital anomalies in babies born to mothers living near landfill sites containing hazardous waste - no apparent dose-response relationship was identified. Studies are generally designed and conducted without con-

sidering variables associated with location and operational controls (Heasman, 1999).

The British Government has initiated a research programme on the health impacts of landfills. The initial phase will be a statistical analysis aimed at detecting any links between the occurrence of congenital anomalies in babies born to mothers living near landfill sites and cancers in people living in the vicinity of landfill sites. The results of this study are expected to be published in the summer 2000. Exposure studies based on the statistical analysis will then be carried out to assess the presence of chemicals in UK landfills, their dispersion and the population's exposure to them.

According to Heasman (1999), potential exposure pathways include 1) landfill gas emissions, 2) airborne dust, and 3) leachate contaminating surface water or groundwater. A conceptual model of the potential exposure pathways that may exist at MSW landfill sites is shown in Figure 6.2 below¹². Originally developed for landfills where a combination of mainly MSW and to a lesser extent hazardous waste has been co-disposed, the conceptual model is equally applicable to MSW landfills. In addition to the potential exposure pathways identified above, threats to human health and the environment may originate from: contamination of surface waters, sediments, and wetlands; and 4) contamination of the landfill site itself (leading to "direct contact" in the figure below).

¹² The figure applies also to the part of incinerated waste, which is disposed of at a landfill.

A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste

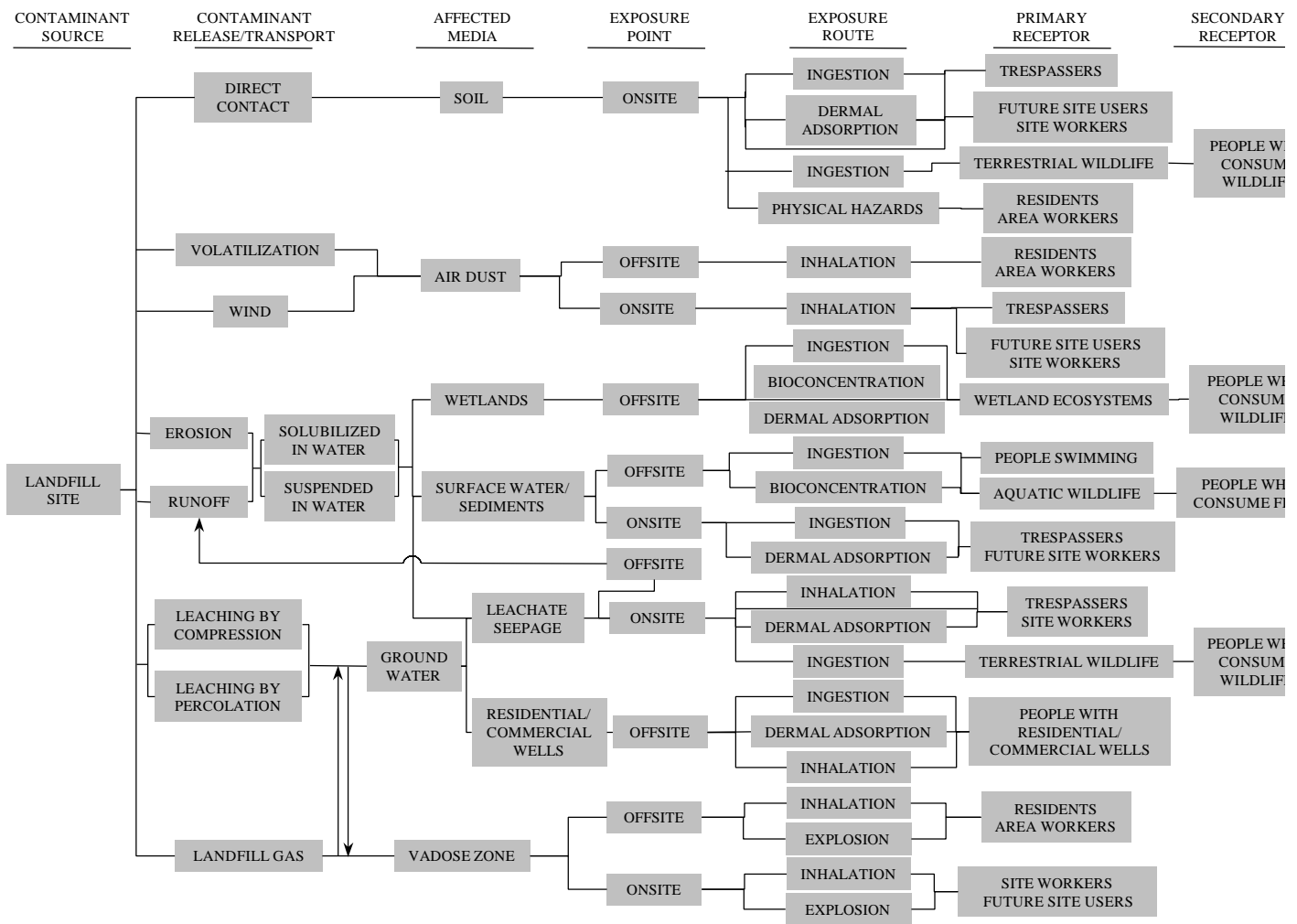


Figure 6.2 Conceptual model of the potential exposure pathways that may exist at MSW landfill sites (Modified from (1994)).

An overview of the emissions to air, water, and soil from landfill disposal and resulting damage is shown in Table 6.1 below. The table gives a general indication of the effect of specific emissions resulting from landfill disposal of MSW, which does not necessarily correspond to the relative importance of these emissions. Thus, the impacts relate to both normal operating conditions as well as to accidental occurrences. The magnitude of the impacts is considered in relation to specific calculation examples.

Other impacts related to landfill disposal of MSW, not shown in Table 6.1, include the displaced impacts resulting from energy recovery and disamenity impacts.

Table 6.1 Overview of knowledge of damage caused by the emissions from landfills, illustrated as dose-response relations.

Emission (dose)	Damage (response)	Medium	Health effects		Lower agri- cultural yield	Forest die-back	Damage to build- ings	Climate effects	Eco- system
			Mortal- ity	Mor- bidity					
CH ₄		Air						*	(())
CO ₂		Air						*	(())
VOCs		Air	(*)		(())				
Dioxins 1)		Air	(*)	((*))					((*))
Dust		Air	((?))	((?))			((?))		
Leachate		Soil and water	((?))	((?))					((?))

Explanation: * Measurable effect, (*) partly measurable effect, ((*)) non-measurable effect, (()) non-measurable but minor effect, ((?)) non-measurable uncertain effect, blank: no known effect.

1) Only when landfill gas is collected and flared or utilised to recover energy.

Land use	Landfill disposal is associated with use of land, which is generally considered to be a negative externality. Potential exposure to MSW at a closed landfill site can affect the possibilities for using the site, and landfill settlement can affect the integrity of buildings erected on the landfill.
Energy recovery	Net energy recovered from landfill disposal of waste displaces environmental impacts associated with the production of energy from conventional sources. Displaced impacts include impacts related to atmospheric emissions, as well as the fossil fuels saved, and residual waste avoided from the energy generation process.
Disamenity	Disamenity impacts result from the operation of the landfill including noise, dust, litter, odour and the presence of vermin.

- Risk of accidents** An accidental or sudden occurrence result in the same outputs as during normal operation, but affects the magnitude of the outputs. Typically, accidents increase emissions, and reduce or prevent energy recovery. The main risks of accidents at landfills relate to biological decomposition processes occurring within the landfill. The risks include surface and underground fires, explosion hazard, and accidental emissions of leachate at containment landfill sites.
- Impact of landfill** The main differences in terms of environmental impacts between landfill sites relate to landfill design and operation, which can influence the emissions to air, soil and water as well as energy recovery. For example, a landfill gas collection and utilisation system reduces the impact on global warming, and via energy recovery also displaces pollution from conventional energy sources.

7 Economic Valuation of Externalities from Incineration

Valuation results for externalities from incineration are presented in this chapter. The chapter is split into four parts. First valuation results of externalities from air emissions are described. Next, valuation of emissions to soil and water is presented. In the following section, external benefits are outlined, and, finally, valuation of disamenity effects is described.

This chapter presents in summary the results from a thorough review of existing literature and research on externalities from incineration of waste. Some areas of externalities are more fully covered than others in this chapter. This reflects the fact that the amount of research and valuation results differs substantially from one area to the other. It is particularly worth noting that the literature on valuation of externalities to soil and receiving water from incineration appears to be very sparse. A full version of the literature review is shown in Appendix IV. Moreover, Appendix III presents the theoretical foundation of the techniques for valuing externalities.

7.1 External Costs of Air Emissions

In this section, valuation results from different studies on air emissions are shown. It appears that the literature falls into two groups. The first group contains valuation studies based on ExternE (1995) in order to quantify impacts. The second group of literature uses the more questionable approach of linked environmental values and thereby obtains quantification of a larger range of pollutants. The main externality costs per kg emission are summarised in Table 7.1.

Table 7.1 Summary of valuation results of air emissions for the studies (EURO per kg emission).

Study (year of valuation)	Study 1 (1998)	Study 2 (1996)	Study 3 (1995)	Study 4 (1993)
Emission type				
CO ₂	-	-	0.04	0.004
PM ₁₀	13.6	28.7	20.5	9.5-12.8
SO ₂	12.2	7.3	2.1	3.1-7.3
NO _x	18.05	18.34	6.0	2.5-4.3
VOC	0.7	2.53	1.4	-
CO	0.00207	-	-	0.007
As	150	999	1,015,735	-
Cd	18.3	81.4	125,370	-
Cr VI	123	819	200,642	-
Ni	2.53	16.8	101,549	-
Dioxins (TEQ)	16,300,000	2,000,000	713,175,937	-
Pb	-	-	34,627	-
Hg	-	-	25,909	-
HCl	-	-	6.1	-
HF	-	-	2,210	-

Notes:

Study 1: Study on Health Risks of Air Pollution from Incinerators. NQ: NO₂ (via xNO₃) + NO₂ (via O₃)

Study 2: Economic Evaluation of the Draft Incineration Directive. VOC: total TOC is indicated. German site used as base case.

Study 3: Miljøkostnader knyttet til ulike typer avfall. Conversions performed with exchange rate in April 2000: 1 NOK = 0.122792 EUR.

Study 4: Cost-Benefit Analysis of the Different Municipal Solid Waste Management Systems: Objectives and Instruments for the Year 2000. Intervals represent the different values for countries.

In the table, studies 1, 2, and 4 are all based on ExternE for conventional pollutants. It should be noted that study 1 only includes health effects whereas the two other studies include other quantifiable effects as well. However, when summing all impacts together, the health costs constitute by far the largest share of the costs. When comparing the results of study 1 and 2 with the results of study 4, the highest estimate should be taken for study 4, because this represents the costs for countries like Germany, Belgium and France, which are the countries for which the other estimates are derived.

The most remarkable difference is that studies 1 and 2 reveal substantially higher costs for NO_x than those from study 4. For SO₂, the costs in study 1 are higher than the others, and for particulates the figure from study 2 is substantially higher than the others. There is not any immediate explanation to these differences. As for CO there appears to be a general agreement that the external costs related to this pollutant are rather small if not negligible.

When looking at the costs of heavy metals (As, Cd, Cr (VI) and Ni) the variation between the costs is much larger. However, it appears that studies 1 and 2,

which are based on epidemiological studies, are in the same order of magnitude. In contrast the results of study 3 based on health indices are much higher. It could be argued that studies 1 and 2 do not look at all health effects from these substances, because not all health effects could be quantified. However, as indicated earlier, the theoretical validity of the valuation approach used in study 3 is questionable, thus bringing into question the robustness of the cost estimates. Therefore, these results should be treated with caution. The same argument prevails for dioxins.

Next, Pb, Hg, HCl, and HF are only priced in study 3. As the unit external costs of these substances are lower than the cost of the other heavy metals in study 3 this underlines the fact expressed in study 1 that these substances are much less toxic than is the case for e.g. As, Cd, Cr (VI) and Ni. However, in this respect it is of course important to know how much of each substance a typical tonne of MSW contains to know whether this substance constitutes a substantial share of the total external costs for a tonne of waste. This is the subject of the calculation examples.

Finally, only studies 3 and 4 contain prices of CO₂. More prices can, however, be seen in Section 8.1. The large variation in the price of CO₂ does reflect the variation seen in other fields dealing with the costs of CO₂. It can also be argued that this cost must in any case be even more uncertain to assess than the costs of those substances previously discussed, because of the nature of the damage connected to global climate change.

7.2 External Costs of Emissions to Water and Soil

This section presents a summary of valuation results from different studies on emissions to water and soil.

Basically, there exist two types of liquid effluents from incineration that potentially is associated with external costs to society. One is associated with the disposal of solid residues, which may contribute to the formation of leachate. The other is wastewater from the incinerator, which contains contaminated liquids/sludges that will be released to the sewer. Both are externalities in the strict sense. However, the latter will not be explicitly discussed in this study, as this is a rather different subject from the study and little information is available. However, this does not mean that wastewater externalities should be ignored.

The quality of the wastewater is monitored and controlled according to conditions laid down in EU directives. Wastewater is often treated at the plant, particularly at modern plants, and/or the local sewage treatment plant prior to discharge to surface water. The sewage treatment plants will charge the waste treatment companies for the wastewater that they release to treatment taking account of the environmental quality of the wastewater (CSERGE et al, 1993 p.79). See also Appendix II (section 1.2. Impacts of Wastewater) and Appendix IV.

Incineration of waste reduces the volume of waste by 90%-95% and solid residues of waste make up 25%-30% of the weight before incineration. However, disposal of the solid residues may contribute to the formation of leachate. The solid residues contain heavy metals and toxins. However, it is uncertain whether these pollutants will be emitted to soil or water, because the residues are disposed to specially lined landfills that are designed to keep the leachate within the site and control any discharge of leachate. Nevertheless, some experts argue that disposal of residues is also associated with emissions to soil and water because in the long run, the lining is likely to rupture. The discrepancy and uncertainty in this area are reflected in the following review of literature on valuation of emission to soil and water.

The few studies available for review illustrate that existing valuation results of emissions to soil and water are scarce. Only very few studies contain valuation results and no study exists, which bases valuation on a damage cost approach.

The Tellus study and the ECON study present valuation results for a number of emissions to soil and water. However, a large proportion of the results is based on a questionable methodology. It is flawed both in its choice of control cost methodology, and in its application of the methodology.

The report "Economic Evaluation of the Draft Incineration Directive" by ETSU for the European Commission, DGXI, 1996 estimates the costs of leachate from solid residues from incineration based on CSERGE et al, 1993. However, the valuation result is not based on a damage cost approach, rather it is based on clean-up costs, which is based neither on WTP nor WTA of individuals. The study estimates the external costs associated with leachate release due to incidents at landfills over a 30-year period in UK at around 1.3 ECU per tonne waste disposed. In this figure it is assumed that leachate from disposal of solid residuals will be equivalent to the leachate from conventional landfills. The estimate is associated with very high uncertainty. ETSU argues that there are several reasons for suspecting that the figure overestimates the external costs, despite of the fact that theoretically clean-up costs only reflect a minimum for the damage value since it would be illogical (or at least not efficient) to pay a greater sum for clean up than the cost of the perceived damage. Firstly, almost all of the incidents reported in UK were regarding older sites that are not very well lined (if at all). Secondly, the figures are not specific to waste of defined and controlled leachability. Thirdly, a large proportion of the reported costs is likely to be internalised. Any water resources that may be affected will be monitored and paid for by the site operators who will pass the costs on to those sending their waste for disposal.

Due to lack of robust results for valuation of emissions to soil and water, it is currently not possible to cite any cost figures. Further, the costs of emissions to soil and water are very site-specific depending on the characteristics of the incinerator and the landfill for disposal of the solid waste. Finally, the estimate will also vary from country to country because of variations in countries' willingness to pay for environmental quality.

If it is absolutely called for to use an estimate for the costs of emissions to soil and water, the most reasonable is to base the estimate on the result from the ETSU study. It is recommended to use the estimate corrected for the fact that app. 30% of waste incinerated is disposed as solid residues and taking account of the fact that the figure study seem to overestimate the cost for leachate.

In order to obtain a solid estimate of the costs of emissions to soil and water more research is needed (see section 8.2 for proposals for further research).

7.3 External Benefits

This section briefly discusses the valuation of external benefits from incineration.

Incineration of waste is not only associated with external costs, but frequently also with both external and internal benefits. For example and most importantly, as already discussed in section 5.1, incineration can be accompanied by recovery of energy.

7.3.1 Displaced Emissions

Energy recovery gives large *internal* benefits for incinerator operators but also *external* benefits for society through displacement of pollution from other energy sources.

Energy recovery from the incineration of waste can consist of both electricity and heat. The recovered electricity and heat in itself is not an externality, since its values directly affect the costs of operating a site and therefore also directly affects the price charged for incineration. Moreover, the financial return from electricity and heat generation is an *internal* benefit to the owner of the incineration plant and therefore not relevant to the estimation of externalities. However, electricity and heat recovery will displace the least profitable form of electricity generation in the electricity system, which means that the recovery of energy will displace pollution from those sources. This gives rise to *external* benefits for society.

In Appendix I (section 1.4), the emissions displaced from the recovery of energy are discussed. The emissions displaced depend on the type of energy displaced in the power generating system. Moreover, the external benefit of displaced emissions depends on the energy source that the energy replaces. If the energy from incineration replaces energy produced by windmills then the external benefits is close to nil, as windmills do not emit pollutants during operation. However, if the energy from incineration replaces energy from conventional energy sources (e.g. fossil fuels) benefits include reduced emissions of CO₂, CO, SO₂, NO_x, particulates (PM₁₀) and others.

When valuing the benefit of displaced emissions, emphasis should be placed on determining/assuming the marginal source of both the heat and the power replaced. In CSERGE *et al* (1993), EC (1996d) and Brisson (1997) the marginal

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

Sometimes it is argued that it is not always correct to assume that the marginal source of power is coal-fired power stations. It is argued that the alternative to using energy from incineration of waste could very well be energy from new, efficient power plants. However, the marginal alternative should always be assessed in the short term from the existing power plants, in which case coal-fired power stations are the right marginal source in most EC countries. The emissions from coal-fired power stations are presented in Appendix XI (section 6.2). Economic valuations of these emissions are not presented in this section, as the values are equal to the ones presented for the same emissions throughout Chapters 7 and 8.

7.3.2 Other Benefits

Incineration of waste produces solid residues, which have proven useful for alternative purposes. Some European countries allow the use of processed bottom ash in construction. For example, Germany, Denmark and the Netherlands use bottom ash as an aggregate in road base material. The bottom ash is therefore a benefit from incineration. However, whether it is purely an internal benefit or partly an external benefit (like the displaced emissions from energy recovery) is a question.

The bottom ash in itself is not an externality, since its value directly affects the costs of operating the incineration. However, net environmental benefits may be associated with the recovery and recycling of materials such as iron and aluminium from bottom ash, and the use of bottom ash as e.g. an aggregate in road construction. The magnitude of these (external) environmental benefits is described in ECOTECH (2000). The external benefit from bottom ash is however not valued in this study. In addition, financial benefits may be associated with bottom ash, although the financial benefits will probably be relatively low.

7.4 External Costs of Disamenity Effects

Waste incineration is associated with several local environmental nuisances such as plant noise, smell, visual intrusion and traffic. Only very few studies have been undertaken that investigate disamenity effects associated with incineration making valuation of disamenity troublesome. However, a number of studies of landfill disamenity effects have been carried out. While there are differences in the types of "aesthetic attributes" and disamenities associated with living close to an incinerator and close to a landfill, there are also obvious similarities. This section will draw extensively on the valuation studies of land-

fill disamenities and therefore the reader is advised to read section 8.4 before continuing below.

Until this point in time, no European studies have been undertaken to investigate the disamenity costs associated with incineration of waste and only very few international studies touching the subject exist. Hence, only a single result of the disamenity costs associated with incineration of waste was found. Due to the origin of the studies, it is not justified to use the results in a European context. Therefore, currently it is not possible to cite any disamenity cost figure for incineration.

Although it is not justified to cite any figure explicitly associated with incineration of waste, it is the impression that the valuations of disamenity from landfill sites and from incinerators might be similar. The comparability is arguable, and no solid evidence for similarities exists. Nevertheless, as a best estimate for the costs of disamenity from incineration at this point in time, it is recommended to apply the estimate of disamenity costs from landfill presented in Section 8.4. Please also refer to Section 8.4 for proposals for further research in this area.

8 Economic Valuation of Externalities from Landfill Disposal

Valuation results for externalities from landfill of waste are shown in this chapter. As for the previous chapter on incineration, this chapter is divided into four sections. First valuation results of externalities from air emissions are described. Next, valuation of emissions to soil and water is presented. In the following section, external benefits are outlined, and, finally, valuations of disamenity effects are described.

All the figures presented in this chapter are obtained from a literature review, as was the case for incineration. Some areas of externalities are more fully covered than others, because the quantity of valuation results in each of the areas is different. Especially the literature covering externalities to soil from landfill disposal is sparse. For a full literature review, the reader should turn to Appendix V in the Appendix Report.

8.1 External Costs of Air Emissions

A summary of valuation results from different studies on air emissions from landfills is shown in Table 8.1.

Table 8.1 Summary of valuation results of air emissions for the studies (EURO per kg emission).

Study (valuation year)	Study 1 (1993)	Study 2 (1995)	Study 3 (1993)
Emission type			
CO ₂	0.002-0.015	0.042	0.004
CH ₄	0.053-0.237	2.223	0.086
VOC	-	1.351	-
N ₂ O	-	-	1.469
VC	-	257.863	-
NO _x	-	6.017	-

Notes:

Study 1: Externalities from Landfill and Incineration. Conversions performed with \pounds -change rate in April 2000: 1 UK \pounds = 1.71233 EUR.

Study 2: Miljøkostnader knyttet til ulike typer avfall. Conversions performed with \pounds -change rate in April 2000: 1 NOK = 0.122792 EUR.

Study 3: Cost-Benefit Analysis of the Different Municipal Solid Waste Management Systems: Objectives and Instruments for the Year 2000

The most important air emissions from landfill disposal are those of CO₂ and CH₄. The table shows a large variation in these costs even within the same study. Study 1 and 3 are both based on Fankhauser (the source quoted in the studies). The results of Study 1 include uncertainty intervals, whereas Study 3 indicates one value for each emission. However, these values are within the intervals indicated for Study 1. It appears that Study 2 using the Norwegian tax on CO₂ has a value somewhat higher than the two others. This study furthermore uses a higher value of the global warming potential of CH₄ and therefore has a substantially higher value for CH₄ than the two other studies. However, according to the earlier recommendations a tax does not necessarily reflect the value of the damage caused by an emission, and it is therefore not recommended to use the results from Study 2. The same argument prevails for the valuation of VOC emissions from Study 2. However, this value is at the same level as values presented in Table 7.1 for air emissions from incineration.

VC is a part of VOC and its specific share will depend on the specific emission of VOC. In Study 2 the valuation of VC reflects the fact that VC normally constitutes a small share of VOC if the two prices should be comparable. However, the same arguments as mentioned earlier imply that the value for VC should not be applied.

The value for NO_x is on the low end compared to the values presented in Table 7.1. For comparison between landfills and incinerators, caution should however be taken to apply the same values for NO_x (as well as other pollutants) consistently (unless there is a specific reason not to do so). The values may furthermore be adjusted to reflect the disposal option (landfill disposal or incineration) by correcting for the population exposed.

No studies have been found valuing other air pollutants from landfill disposal than those mentioned above. However, as mentioned in Section 6 there are also some emissions connected to flaring the landfill gas. It is most likely due to the minor importance of these that they have not been valued in any of the studies shown.

8.2 External Costs of Emissions to Water and Soil

This section presents a summary of valuation results from different studies on emissions to soil and water from landfill disposal.

The infiltration of precipitation and surface water into landfills coupled with the biochemical and physical breakdown of waste produce a liquor or leachate with a high organic and inorganic content. The leachate causes various adverse impacts. The focus of this section is valuation of these impacts.

Only few studies contain valuation results for emissions to soil and water and only a small part of these base valuation on a damage cost approach. The large majority of the study results is based on approximate valuation approaches such as control cost and linked environmental values. The results of the studies are summarised below.

Table 8.2 Summary of valuation results of emissions to soil and water (EURO).

Study	Study 1 (1993) per tonne waste landfilled	Study 2 (1997) per tonne waste landfilled	Study 3 (1995) per kg emission to water	Study 3 (1995) per kg emission to soil
Leachate	0.77 (0 - 1.54)	0 - 1.09	-	-
Lead (Pb)	-	-	178	5
Cadmium (Cd)	-	-	622	1,514
Mercury (Hg)	-	-	1,022	37
Dioxins	-	-	62,824,889	n.a.
Antimony (Sb)	-	-	121,366	121,366
Arsenic (As)	-	-	308	12
Barium (Ba)	-	-	31	37
Beryllium (Be)	-	-	44,928	44,928
Copper (Cu)	-	-	5	1
Chromium (Cr) ¹⁾	-	-	17,479	320
Nickel (Ni)	-	-	12	4
Selenium (Se)	-	-	16,125	16,125
Zinc (Zn)	-	-	1	1

Notes:

1) Split 50/1 between Cr III and Cr VI.

Study 1: Externalities from Landfill and Incineration. Based on cleanup cost. Conversions performed with exchange rate in April 2000: 1 £ = 1,71233 EURO.

Study 2: Waste not, Want not: the Private and Social Costs of Waste-to-Energy Production. The emissions considered consist of As, Cd, Cr (type not indicated in source), Cu, Ni, Pb, and Hg. Conversions performed with exchange rate in April 2000: 1 £ = 1,11495 EURO.

Study 3: Miljøkostnader knyttet til ulike typer avfall. Conversions performed with exchange rate in April 2000: 1 NOK = 0.122792 EURO.

Study 1 bases valuation on cleanup-cost, which entails that all impacts associated with leachate are included in the study. Study 2 uses a marginal damage cost approach for valuing leachate. The leachate in the study is assumed to consist of As, Cd, Cr, Cu, Ni, Pb, and Hg, which are therefore the emissions taken into consideration. The marginal damage functions used to estimate the damage of the emissions are limited to include only mortality and morbidity effects. Study 3 uses an approach for valuation based on a control cost methodology coupled with linked environmental values or indices, where costs are quantified on the basis of the cost of one pollutant coupled with linked environmental values. Both environmental and health effects are covered by the study.

The results of study 1 in comparison with the results of study 2, show that the estimate of leachate costs per tonne waste incinerated is very close in the two cases, despite the fact that the two studies use very different methodologies for deriving the estimates. Therefore, it is tempting to conclude that since the two estimates are very close, a mean of the two estimates of costs per tonne waste landfilled is a satisfactory and solid estimate. However, due to the fact that the

results from study 1 is not based on a damage cost approach, it is not justifiable to verify the results of study 2 by the results of study 1.

The result from study 3 cannot be compared with the results from the other studies due to the fact that results are presented as costs per emission, whereas study 1 and 2 only present results as costs per tonne waste landfilled. The ECON study presents valuation results for a number of emissions to soil and water, but as already mentioned the results are based on a dubious methodology. Therefore, these results should only be used with great care, if at all.

It is assessed that it is currently not possible to cite any cost figures for emissions to soil and water. However, if it is necessary to use an estimate for the costs of emissions to soil and water, it is recommended to use the mean value of the two mean values from study 1 and 2.

In order to obtain a solid estimate of the costs of emissions to soil and water more research is needed. The available information of leachate damage costs is scarce and insufficient. Better risk analysis coupled with an economic valuation exercise (based on dose-response approach) is needed.

The risk assessment of leachate releases from landfill sites is not adequate. Therefore, proper dose-response valuations have still not been undertaken for valuation of emissions to soil and water. So far, only few dose-response valuations have been undertaken - presenting results aggregated for all emissions (Miranda and Hale, 1997). Better understanding of the pollution pathways of leachate and of the relationship between exposure and damage is called for (dose-response), to enable proper valuation of the adverse impacts using the impact pathway approach. One needs to know more about the effects caused by old landfills closed long ago, e.g. the effects on long-term accumulation of persistent pollutants such as heavy metals.

As an alternative to dose-response valuations other valuation approaches can be undertaken to obtain results for the costs of emissions to soil and water. Clean-up costs and avoidance costs approaches might be more suitable and justified as long as the all effects of leachate in short and long term is not known, although the valuation based on these techniques do not reflect individuals' willingness to pay.

Valuation techniques that are not based on the fundamental principle of using individuals' willingness to pay in the monetary valuation estimation do not offer true welfare measures but only crude approximations. The clean-up cost approach does not come without flaws and problems, but it is an important approximate valuation technique, since in many contexts it may be difficult or impossible to find more direct measures of damage. Strictly speaking, the costs of cleaning up are not a substitute for damage costs and the costs need to be used with caution if they are applied as measures of damage.

8.3 External Benefits

This section briefly discusses the external benefits from landfill disposal of waste.

8.3.1 Displaced Emissions

In some landfill sites the landfill gas produced is collected and used as an energy source for production of electricity and/or heat. See section 6.1.

Similar to the energy recovered from incineration, the landfill gas produced and used as an energy source for electricity and heat in itself is not an externality, since its values directly affect the costs of operating a site. However, the energy produced from the landfill gas will displace the least profitable form of energy production, which means that the recovery of energy will displace pollution from those sources. Therefore, the situation is exactly similar to the situation of displaced emissions from energy recovery discussed in relation to incineration. See section 7.3.1.

8.3.2 Other Benefits

Other benefits from landfilling are not considered in the waste studies reviewed. However, it should at least be mentioned that landfill generates benefits from land reclamation.

Land is reclaimed through infilling, permitting the land to be used for purposes it would otherwise not have been able to support. It is, however, unclear whether land reclamation is an internal or an external benefit. It depends on who gets the property rights of reclaimed land. If the site operator gets the property rights, the benefit is internal, whereas it is external if society (or another agent) gets the ownership of the land. No information is available on this subject and therefore it is not considered further in this study.

8.4 External Costs of Disamenity Effects

In this section, a summary of valuation results from different studies on disamenity from landfills is shown. A number of studies investigating disamenities associated with landfill operation has been carried out in the US while only two single studies to this date in time have been carried out in a European context.

Nuisances caused by landfills include odour, flies, seagulls, wind-blown litter, noise, visual intrusion, and traffic. Moreover, disamenities or disamenity effects include deterioration in "aesthetic attributes" associated with environmental goods. The aesthetic attributes include deteriorations in taste, odour, appearance, or visibility. In short, these costs are determined by how the senses are affected and how individual welfare is changed as a result. This class of costs is unique in the sense that the focus is on the sensory experience and not on a physical or material effect. Therefore, there is a clear-cut conceptual distinction

between aesthetic costs and physical or material costs. However, despite of the distinction, the valuation of disamenity is often mixed with the costs of impacts from air pollution such as health effects and risks. The reason is that a policy that improves air quality, for example, might simultaneously improve visibility and reduce mortality risks associated with airborne contaminants. In a willingness to pay study, it often proves very difficult or impossible to separately quantify and value improvements of the different effects.

Three studies containing valuation results for disamenity associated with landfilling of waste were found. To our knowledge Garrod and Willis, 1998 and an ExternE study¹³ are the only studies undertaken in a European context, which explicitly focuses on estimating the costs of disamenity associated with landfills. The Brisson and Pearce, 1995 study summarises and reviews the result of disamenity effect studies in the US. Based on the review, Brisson and Pearce derive a best estimate for the costs of disamenity.

Unfortunately, the best estimates from the two studies are not presented in a form that allow for explicit comparisons.

The Garrod and Willis study bases valuation on a stated preference approach. It concludes that the marginal willingness to pay to reduce the number of days suffering from dust and windblown litter from the landfill is around 14-17 pence per day. Further the marginal willingness to pay to reduce the number of days when the landfill can be smelt from their home is around 9-14 pence per day. The majority of respondents are unwilling to pay for any improvements, regarding the current level of disamenity negligible, while the minority is willing to pay a relatively small amount.

Brisson and Pearce base the best estimate on a regression analysis of the results from the studies reviewed, which all base valuations on hedonic pricing or contingent valuation. The regression analysis suggests a willingness to pay equal to a maximum decline in house prices of 12.8% occurring at the site of the waste disposal facility, and that the effect on house prices will have fallen to zero at a distance of 3.4 miles from the facility.

It is not possible to compare the estimates of these two studies without making a number of assumptions. For example, one needs to assume the number of days that the surrounding dwellings are affected by windblown litter and dust and smell from the landfill, the average house price, the number of dwellings affected and the distribution of the dwellings. All in all, the many and vital assumptions make it very difficult to compare the results. In fact, it is not justifiable to verify the study results by mutual comparisons.

The results of the Brisson and Pearce study are considered relevant for valuing the disamenity costs of landfills, but some caution is called for because of the geographical origin of the studies. The Garrod and Willis study also contains relevant results, although the study is based on limited data and investigates

¹³ ExternE Externalities of Energy, Vol 9. The evaluation of disamenities of waste disposal site.

only disamenities for well-established landfill at a very specific site in the UK. It is not the impression that any of the valuation results presented above are solid enough to be adopted as a robust estimate for disamenity cost for landfill. However, if it is necessary to use an estimate for the costs of disamenity from landfills, it is recommended to use the results of both studies depending on the situation at hand.

As already indicated above, there is plenty of room for improvements of valuation of landfill disamenities. Firstly, the existing studies are generally quite old (from 5-20 years). Public preferences change over time and it is likely that people's concern for the environment has increased over the last decades. Therefore, it is essential that new studies investigating people's preference with respect to environmental disamenities are undertaken.

Secondly, there is a huge deficiency of studies undertaken in a European context, while US studies are well represented. Differences in WTP for environmental attributes are known to exist across countries. Moreover, it is not known if the valuations relevant to US sites are applicable to European sites and therefore there is a practical problem of benefits transfer. In addition, the public in some countries appears to be more accepting of incinerators than in other countries – which might reflect differences in disamenities or perceived health risks.

More European WTP studies of landfill disamenities are needed. Appropriate procedures for carrying out the valuation include hedonic pricing and contingent valuation. However when commissioning such studies, the potential for benefits transfer should be given serious thought. The values obtained from hedonic property price studies and contingent valuation studies are generally values for a composite of attributes. Therefore, the values cannot subsequently be disaggregated to allow for transfer to another study site, which might have a different combination of attributes. In addition, the values thus obtained often encompass other attributes than just disamenities, such as e.g. health effects. Valuation studies which allow estimation of values for individual attributes, e.g. visual intrusion, wind-blown litter, odour, etc., would appear to offer greater potential for transfer of estimates between study sites. Stated preference techniques which offer this possibility, including e.g. choice experiments, contingent ranking and contingent rating, have been developed within the fields of market research and transport economics, and have in recent years also been adopted by environmental economists. A pilot project estimating the disamenity effects from a landfill (using choice experiments) has been conducted by Garrod & Willis (1998) as reported above.

In summary, three further European research within waste disamenity could be carried out along one of the below lines:

- 1 To do valuation studies in all EU countries (which would be expensive);
- 2 To do some valuation studies alongside survey of citizens in all EU countries;

- 3 To take as fact that there are different views in different countries and to do valuation studies in one or two countries where people are hostile to landfill disposal / incineration, and one or two countries where people are less so.

9 Examples of Externality Cost Calculations

Chapters 5-8 dealt with emissions and impacts of the externalities associated with landfill disposal and incineration of waste and presented economic valuation estimates of these externalities. This chapter presents examples of externality cost calculations for incineration and landfill disposal of waste under different assumptions. The calculations are based on the information presented in the other parts of the report and the Appendix Report.

The aim of the examples is to quantify the main externalities according to typical scenarios for landfill disposal and incineration of waste in terms of physical impacts and monetary values. The examples have been designed to reflect respectively old obsolete and new modern waste disposal plants for both incineration and landfill disposal. The examples include the external costs of the following components:

- greenhouse gases;
- conventional air pollutants and some airborne toxic substances;
- leachate; and
- disamenity effects.

The examples do not include external costs of pollution and accidents associated with transportation of waste to incineration plants and landfill sites nor do they include any other externalities associated with transportation. The calculations also include external benefits in the form of displaced emissions from energy recovery.

It should be mentioned that the aim has been to put forward all available information. The main part of the costs presented in the following is hence marginal external costs, whereas the external costs of disamenity are average costs. It could be argued that the correct values to add are either marginal costs or average costs. The marginal disamenity costs are zero, which is highly different from the numbers presented in the following sections. The reader should keep this in mind.

It should be emphasised that several reasons exist why the results are associated with considerable uncertainty, and the results should therefore be treated with great caution. Firstly, emission data are based on several assumptions and they are especially uncertain in respect to emissions with no proposed EU limits. Secondly, not all emissions are included in the calculations. Several heavy me-

als and toxic substances are omitted simply because no reliable estimates exist for these emissions. It is, however, assessed that the most important air emissions are included and priced, emissions to soil and water are not priced directly but valued as a one component with a cost per tonne of waste. Thirdly, the valuation estimates in general - and specifically the valuation estimates for leachate and disamenity - are highly uncertain.

Because of the considerable uncertainty, the results presented below are generally expressed in terms of a range of values. The variation in the results is based on different assumptions concerning the physical emissions and the monetary valuation estimates.

Appendix VI gives a more detailed explanation of the underlying assumptions and background emission and valuation data and the reader should refer to the appendix for more details.

9.1 Incineration

Three examples have been defined for incineration of waste. They reflect different technological standards and level of energy recovery. The examples are as follows:

- **I1.** The incineration plant fulfils the proposed directive on incineration of waste (Common Position (2000/C 25/02)). Energy recovered will generate electricity and heat (CHP), which normally implies a high recovery percentage. This percentage is assumed to be 83%.
- **I2.** The incineration plant fulfils the existing directive on incineration of waste (89/369/EEC). Energy recovered will generate electricity only, which normally implies a lower recovery percentage. This percentage is assumed to be 25%.
- **I3.** The incineration plant does not fulfil the existing directive. The flue gas cleaning technology is an electrostatic precipitator. There is no energy recovery.

All examples cover the incineration of MSW as defined in the introduction of this report. All figures are presented as costs per tonne of waste. This is the natural way to present the results for the variable externalities. However, when looking at the disamenity effects of incineration this way of presenting the figures implies that assumptions have been made regarding the capacity of the incineration plant and the number of households surrounding the plant. These assumptions are shown in Appendix VI

The marginal energy source for electricity, which is replaced by electricity produced at the incineration plant, is assumed being a coal-fired power plant. The emission data from the power plant are the lifetime emissions, which implies that emissions from mining and transport are included. This somewhat overestimates the benefit from waste incineration, as the emissions from transport-

tion of the waste are not included on the cost side. However, the largest share of the emission from the coal power plant is related to the actual operation of the power plant. The marginal energy source for heat, which is replaced by heat produced at the incineration plant, is assumed being a coal-fired district heating system.

Coal is considered as giving rise to the largest benefit in terms of displaced emissions. As sensitivity case, oil has been chosen as the displaced energy source. The emission factors of the electricity produced by oil only include production of the power plant as opposed to those of the coal power plant.

As regards heating, it may be more likely that the generated heat would replace heat produced by oil, which means that a more realistic situation would be a combination of the two calculations presented here.

Wastewater from the incinerator contains contaminated liquids/sludges that will be released to the sewer. The external costs from liquids/sludges from incineration are not included in the calculations¹⁴.

In each of the examples, it is assumed that 30% of the waste incinerated (by weight) remains after the incineration process, and that this waste is brought to a landfill with leachate management. This gives rise to external costs of leachate from the residual product. Table 9.1 shows the results of the examples.

¹⁴ Often sewage treatment plants charge the waste treatment companies for the wastewater that they release to treatment taking account at least partly the environmental quality of the wastewater. If the price paid is equal to the external costs the externality is fully internalised. However, if the price does not cover the external costs, wastewater from incineration is still an externality for society.

Table 9.1 Summary of externality costs for incineration of waste in examples I1, I2 and I3 (EURO/tonne waste incinerated)

Example no.	I1	I2	I3
Impact			
Global warming	0.8 (0.5 – 1.0)	0.8 (0.5 – 1.0)	0.8 (0.5 – 1.0)
Damage from air pollution ¹⁾	20 (5 – 27)	50 (15 – 72)	69 (20 – 108)
Damage from leachate	0 (0 – 0.3)	0 (0 – 0.3)	0 (0 – 0.3)
Disamenity	8 (4 – 14)	8 (4 – 14)	8 (4 – 14)
Total external costs	28 (10 – 43)	58 (20 – 88)	77 (25 – 124)
Pollution displacement ¹⁾	-71 (-115 – -19)	-21 (-29 – -4)	0 (-)
Net external costs	-43 (-72 – -9)	37 (16 – 84)	77 (25 – 124)

Note: Low, high and best estimate values for both emissions and prices were used. The low end of the interval is obtained by using low values of each estimate and the high values are obtained by using high values of each estimate. This will overestimate the size of the interval. This approach is also used for pollution displacement, which has the adverse effect.

1) The main part of these costs/benefits is related to NQ and SO₂.

The calculations show that in all examples the largest external costs are the costs of air pollution. Second largest are the disamenity costs, and the leachate costs are zero or very small. The external costs are (not surprisingly) lowest in example I1 with the most modern incineration plant and highest for the old incineration plant in example I3.

The external benefit has a lot of influence on the results. The calculations show that there is a net external benefit associated with incineration of waste on an incineration plant fulfilling the proposed directive on waste (I1) and recovering the maximum amount of energy. If such a plant was instead recovering electricity only there would be a net external cost associated with the incineration of waste. Again, it should be recalled that the example may be optimistic in regards to the displacement of heat produced by coal and that it may be more realistic, if the heat replaces heat produced by another source (see also Table 9.2).

Looking at a plant fulfilling the existing emission norms (I2) the same relationship prevails. However, the external costs are larger for such a plant and in the example with recovery of electricity only, the incineration is associated with a net external cost.

Finally, an incineration plant, which does not fulfil the existing emission norms, will independently of energy recovery, be associated with a net external cost.

The disamenity costs constitute a substantial share of the total external costs. Two things should be noted in this connection. First, the cost is obtained using US study results. This is done because of the lack of European studies. There is thus no validation of whether the results do in fact apply to European conditions. Secondly, the disamenity value may include part of the other externalities. For instance, the perceived disamenity may take into account the actual increased health risk from air pollution, which is already accounted for in the cost of the air pollution externalities. If this is the case, the disamenity value is overestimated.

Table 9.2 shows the result of the examples if the displaced energy is produced with oil instead of coal. This implies lower external benefits, which result in higher net external costs. However, the result of I1 is still a net external benefit (though much smaller) and the results of I2 and I3 are likewise still net external costs. Hence, the conclusions are not altered in the sensitivity case with the assumption that oil is the marginal energy source, but the benefit in I1 is much closer to zero than before.

Table 9.2 Externality costs for incineration of waste in examples I1, I2 and I3 with oil as alternative energy source (EURO/tonne waste incinerated)

Example no.	I1	I2	I3
Impact			
Total external costs	28	58	77
Pollution displacement	-37	-14	0
Net external costs	-9	44	77

Generally, a very uncertain aspect of the calculations is the cost of leachate. In the examples, this cost is zero or very low and, hence, almost negligible compared to the other externalities. The figure is obtained on the basis of very few studies of valuation of leachate. Furthermore, the values are obtained mainly from clean-up costs and it may therefore be the case that the value is underestimated. In the next section, the emissions of leachate are described a bit further.

The examples show that from a purely environmental point of view I1 is better than I2, which is again better than I3. However, maybe the associated costs of leachate are underestimated and this may alter the results. It should furthermore be recalled that in a cost-benefit evaluation the internal costs have to be taken into account as well. Hence, the external benefit or decreased external costs obtained by increasing the environmental standards have to be measured against the internal or financial costs of the option.

9.2 Landfill Disposal

Two examples have been defined for landfill disposal of waste. They reflect different technological standards and level of energy recovery. The examples are as follows:

- **L1.** The landfill is a modern containment landfill that fulfils the demands of the newest directive (EC/31/1999). The landfill has a leachate collection and treatment system. Further, the landfill gas is collected to generate electricity and heat (CHP).
- **L2.** The landfill is an old site without a liner and landfill gas is not collected.

The examples cover the disposal of MSW as defined in the introduction of this report. All figures are presented as costs per tonne of waste. Again, for the disamenity effects this implies that assumptions have been made regarding the capacity of the landfill site and the number of households surrounding the plant. These assumptions are shown in Appendix VI

The marginal energy source for L1 is assumed being a coal-fires power plant and a coal fired district heating system. As sensitivity case oil has been chosen as the displaced energy source.

Table 9.3 shows the results of the examples.

Table 9.3 Summary of external costs for landfill disposal of waste in examples L1 and L2 (EURO/tonne waste disposed at landfill)

Example no. Impact	L1	L2
Global warming ¹⁾	5 (1 – 14)	8 (2 – 23)
Damage from air pollution	0.1 (0.02 – 0.2)	0 (-)
Damage from leachate	0 (0 – 1)	1.5 (1 – 2)
Disamenity	10 (6 – 19)	10 (6 – 19)
Total external costs	15 (7 – 34)	20 (9 – 44)
Pollution displacement ²⁾	-4 (-10 – -1)	0 (-)
Net external costs	11 (6 – 24)	20 (9 – 44)

Note: Low, high and best estimate values for both emissions and valuations were used. The low end of the interval is obtained by using low values of each estimate and the high values are obtained by using high values of each estimate. This will overestimate the size of the interval.

1) The main part of these costs is related to CH₄.

2) The main part of these benefits is related to NO_x and SO₂.

The results show that the largest external costs of landfill disposal of waste are the disamenity costs. The second largest costs are those of global warming emissions especially methane. Leachate costs are zero for L1 and constitute a small share of the costs for L2 even though these costs are estimated to be larger than for incineration.

Again, the external costs are highest in example L1 and due to the external benefit of displaced emissions, the difference between L1 and L2 in the net result becomes bigger.

The main concerns of the results are the costs of disamenity and leachate. The disamenity costs are based on US studies and might therefore not be applicable to European conditions. This costs component should therefore be treated with care. The cost of leachate is as for incineration based on very few studies and the cost does not price each of the substances in the leachate but values emission of leachate as one component based on clean-up costs. It is therefore uncertain whether this value reflects the true external costs. It is nevertheless assessed that leachate is almost non-existing for landfills with leachate management. This is described in more detail in Appendix XI.

Table 9.4 shows the results of the examples when the replaced energy source is oil instead of coal. Conclusions to be drawn are only altered a little by this change.

Table 9.4 External costs for landfill disposal of waste in examples L1 and L2 with oil as alternative energy source (EURO/tonne waste disposed at landfill)

Example no. Impact	L1	L2
Total external costs	16	20
Pollution displacement	-3	0
Net external costs	13	20

The examples show that from a purely environmental point of view L1 is better than L2. Again, the external benefit or decreased external costs obtained by increasing the environmental standards have to be measured against the internal or financial costs of the option.

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