Valuation of the external costs and benefits to health and environment of waste management options

Final report for Defra by Enviros Consulting Limited in association with EFTEC

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Authors

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In spite of these various contributions the conclusions in the report are the views of the authors, and we remain responsible for any remaining oversights or omissions.

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Executive summary

This report presents the findings of a study conducted by Enviros Consulting in conjunction with Economics for the Environment Consultancy (eftec) to provide an assessment of the external costs and benefits to health and the environment of waste management options in monetary terms.

The study has been commissioned by Defra to support the work of an Inter Departmental Group on waste management.[1] The aim of the group is to examine, in scientific and economic terms, the health and environmental impacts of different waste management options.

The terms of reference for this study were to:

1. Review the literature on impacts from waste management facilities and determine economic values; and

2. Integrate these findings with a separately commissioned scientific study.

This economic study completes the second stage in a two stage process. The first stage has assessed scientific evidence of the physical health and environmental effects of options to manage municipal solid waste (MSW) and similar wastes. For convenience the first stage work is referred to as the “scientific study”, although strictly speaking it is simply a study of physical impacts. In this second stage, monetary values for the physical impacts estimated in the first stage are reviewed and aggregated to the level of a unit of treated/disposed waste (e.g. £/tonne of waste). This convention makes it possible to compare, in common units, the relative health and environmental effects of alternative methods of managing and disposing MSW or similar waste.

Any comparisons are subject to significant uncertainty. Nevertheless it is important to bring together the available literature on values and their application to MSW management processes based on the available information. Such analysis should allow some comparison of the external costs of waste management to be made. We have thus undertaken a comprehensive review of the secondary and primary literature on the economic valuation of health and environmental externalities.

The following tables summarise the results from the most relevant primary studies. The recommended estimates in these tables are to be treated with caution, as there are still large uncertainties over the values. Therefore, we strongly recommend that when comparing the economic impacts from MSW management options, a full sensitivity analysis is conducted to explore implications of the uncertainties in these values.

1 The Inter Departmental Group was set up in response to a commitment in the PBR 2002 to review the environmental and health effects of all waste management and disposal options.

Health impacts

Table E.1 shows the summary values attributable to health impacts. These are obtained from the primary studies identified in Section 4. Recommended estimates are provided to be consistent with the outputs of the scientific study. In Table E.1 both ranges and point estimates are given. The point estimates do not necessarily imply greater accuracy, but reflect the lack of range provided in the underlying literature.

Table E.1 Summary values for health impacts (£, 2003)

	Recommended Estimates	Initial Range / Figure
Deaths brought forward	***3,100 – 110,000***	***3,100 – 1,600,000***
Respiratory admissions (Casualty & Hospitalisation)	***550 – 1,260***	***550 - 1,260***
Cardiovascular hospital admissions	***1,337 – 4,011***	***2,674***
Cancers		
Lung	1,480,000	53,000 – 1,480,000
Leukaemia	2,260,000	199,000 – 2,260,000
Haemangiosarcoma	cannot recommend	no value available
All cancers	***1,150,000 – 2,260,000***	***53,000 – 2,260,000***
Birth defects		
Neural tube defects	246,000 - 422,000	246,000 – 422,000
Cardiovascular defects	224,000 - 423,000	224,000 – 423,000
Hypospadias and epispadias	cannot recommend	no value available
Abdominal wall defects	148,000	148,000
Gastroschisis and exomphalos	90,500	90,500
Low birth weight	cannot recommend	no value available
Very low birth weight	cannot recommend	no value available
All birth defects	***90,500 - 423,000***	***90,500 – 423,000***

Air pollution impacts

As described in Section 4.2 because no dose response functions for environmental impacts are included in the scientific study, it is not possible to apply monetary values to the final environmental impacts of the releases from MSW management facilities. Rather we have had to use estimates of the value of impacts on per unit of pollutant. These estimates are based on an independently commissioned modelling exercise for landfill and incineration facilities using analysis from the forthcoming study for Defra evaluating policies included in the Air

Quality Strategy.[2] As such the values are based on independent research. The summary results are presented in Tables E.2 and E.3. Some of the health impacts in Tables E.2 and E.3 are those that are reported separately in table E.1. Therefore, in order to avoid double counting, as far as possible, health and environmental effects in these tables have been separated out. If the per unit of pollutant estimates are used in conjunction with the estimates of unit health impacts and dose-response functions, then only the environmental values reported in Tables E.2 and E.3 should be used.

Table E.2 Summary values for key pollutants from UK landfill (£/tonne pollutant, £2003)

	Coverage	Central Low	Central High
PM10	***Health effects only***	***161***	***1,025***
SO_2		***643***	***2,941***
Of which health	Health effects only	418	2,716
Of which materials	Materials	225	225
NO_x	***Health effects from secondary pollutants, but excluding ozone***	***154***	***977***
VOC		***263***	***665***
Of which health	Health effects, including ozone	3	405
Of which crops	Crop damage, including ozone	260	260
CH_4	***Climate change only***	***158***	***630***
CO_2	***Climate change only***	***9.5***	***38***

2 AEA Technology, Evaluation of the Air Quality Strategy – a study for Defra (Environment Policy Economics) forthcoming.

Table E.3 Summary values for key pollutants from UK incineration (£/tonne pollutant, £2003)

	Coverage	Central Low	Central High
PM10	***Health effects only***	***6,119***	***39,245***
SO_2		***643***	***2,941***
Of which health	Health effects only	418	2,716
Of which materials	Materials	225	225
NO_x	***Health effects from secondary pollutants, but excluding ozone***	***154***	***977***
VOC		***263***	***665***
Of which health	Health effects, including ozone	3	405
Of which crops	Crop damage, including ozone	260	260
CH_4	***Climate change only***	***158***	***630***
CO_2	***Climate change only***	***9.5***	***38***

Water related impacts

The economic valuation literature for the water environment provides monetary estimates for some potential impacts but does not link these to pollutants; dose response functions for the water environment being especially difficult to generalise. Thus, it is difficult to recommend valuation estimates without further analysis of the impacts from MSW waste management on the water environment. Some studies have been undertaken with respect to leachate but these have been of limited scope and the conclusions are not transferable to the UK.

Disamenity impacts

Table E.4 summarises the value of disamenity impacts from MSW landfill and incineration facilities in the UK. As noted in the footnote to the table however, only the disamenity effects of landfill are suitable for application to UK MSW management processes. This clearly makes a comparison of landfill and incineration from a disamenity point of view incomplete.

Table E.4 Summary values of disamenity impacts from primary studies

Impact from Scientific Study	Primary Economic Study Reference	Units	Low	High	Best estimate
			£,2003		
Landfill (hedonic pricing method)	Defra (2003)	£ per facility	551,000	789,000	670,000
		£ per tonne of waste	2.50	3.59	***2.50 – 3.59***
Incinerator (hedonic pricing method)	Kiel and McClain (1995)	£ per facility	Not available	Not available	7.8million
		£ per tonne of waste	Not available	Not available	***21*** [(a)]

Note: [(a)] Not recommended for benefits transfer to a UK context.

Conclusions

The scientific and economic thinking underlying the values provided in this report is constantly evolving. Hence, these values should be seen as a snapshot of the best estimates available at this time of the external costs and benefits to health resulting from waste management facilities in the UK. Whilst the coverage of impacts in the economic literature is reasonably comprehensive, including health and environmental impacts, there are still large uncertainties in the range of values provided. This is not surprising given the nature of benefits transfer and the precision demanded when examining impacts from specific types of facilities such as incinerators and landfills, where the physical risks to health and the environment are generally low and difficult to quantify. Research valuing health and environmental externalities is constantly being updated, and as this understanding improves and greater precision is obtained on the value of these impacts, cost benefit analysis will become more relevant for waste policy decision-making.

1. Introduction

This report presents the findings of the work undertaken by Enviros Consulting Ltd in association with Economics for the Environment Consultancy Ltd (eftec) to assess the external costs and benefits of waste management options.

The study has been commissioned by Defra to support the work of an Inter Departmental Group on waste management.[3] The aim of the group is to examine in scientific and economic terms the health and environmental impacts of different waste management options.

This economic study completes the second stage in a two stage process. The first stage has assessed scientific evidence of the physical health and environmental effects of options to manage municipal solid waste (MSW) and similar wastes. For convenience the first stage work is referred to as the "scientific study", although strictly speaking it is simply a study of physical impacts. The scientific study was prepared by Enviros Consulting Ltd together with the University of Birmingham, Risk Policy Analysts and the Open University and was published in May 2004 (Enviros, 2004).

In this, second stage, monetary values for the physical impacts estimated in the first stage are reviewed and expressed per unit of treated/disposed waste (e.g. £/tonne of waste). This makes it possible to compare, in common units, the relative health and environmental effects of alternative methods of managing and disposing MSW or similar waste.

1.1 Scope of the economic study

This study is a review of the economic valuation literature in order to facilitate monetary assessment of the health and environmental impacts of managing MSW and similar wastes. No primary research has been undertaken.

The scope of the analysis in both the scientific and economic studies is restricted to the health and environmental effects caused directly by the waste facilities. Impacts occurring during other phases of the life cycle of the processing and treatment of waste materials are not included in this research. In particular, the research does not include the effects of emissions from the transport of materials and waste.

The literature review has taken as its starting point studies that have attempted to collate estimates of the externalities of waste management. These studies are referred to as secondary studies because they use valuation data derived from other, often primary, research studies. This has, in turn, led on to a review of the primary studies, as well as secondary studies that are focused on the valuation of air pollution, such as ExternE and BeTa.

3 The Inter Departmental Group was set up in response to a commitment in the PBR 2002 to review the environmental and health effects of all waste management and disposal options.

For health effects, this study reports the estimated values for different types of health impacts. For environmental effects, estimated values per unit of pollutant (tonne or kg in most cases) and per facility (for disamenity effects) are reported since the available information was not sufficient for the scientific study to quantify the impacts on environmental receptors arising from the MSW management facilities.

With regard to disamenity effects, the scientific study does provide evidence of incremental noise levels close to certain MSW management facilities (composting, material recycling facility, landfill and gasification/pyrolysis) as well as the number of noise and odour related complaints from landfill sites. Whilst changes in noise levels and complaints do impact on human welfare, the levels identified as arising from the above MSW facilities are generally low compared to background levels.

As explained in the scientific study, the data on complaints do not distinguish between several people each registering a complaint and several complaints being made by one person. We are not aware of any work that attempts to value a complaint. There are also no studies that measure the economic cost of relatively small increases in noise levels compared to background. Most literature on noise impacts refers to larger sources of noise such as motorways and airports. For these reasons we have taken results from valuation studies that capture the total disamenity effects around MSW facilities.

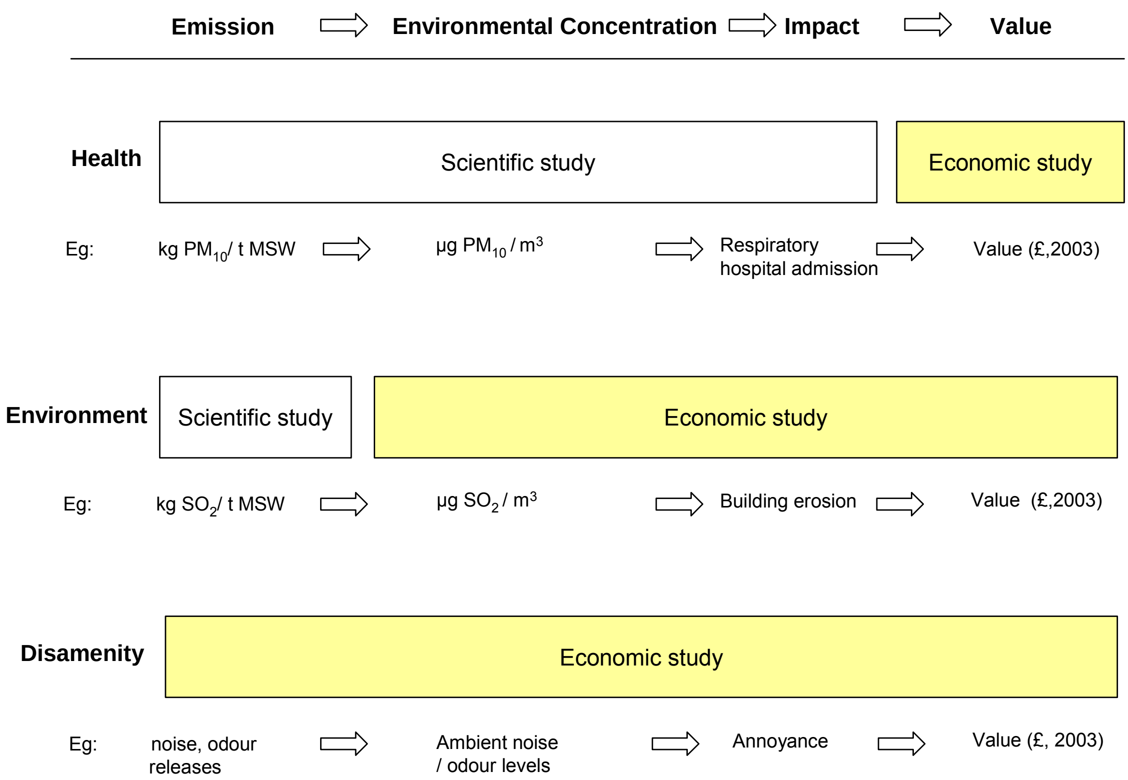
Another impact that is not explicitly quantified in this study is the consumption of land available for landfill, the implication being that future users of the land are being deprived of making use of the asset by the current users.

Such an externality would exist where the social costs of land acquisition are higher than the prices paid for the land by landfill developers. From a qualitative perspective, one can say that in perfect markets the scarcity of a natural resource would be built into its price and hence social costs would equal private costs. Deviation from this position would be caused by a failure in the market for land (for example through oligopolistic behaviour on the part of landfill developers) or by a failure of land use planning policies. The landfill industry, however, is widely regarded as highly competitive, and we are unaware of any serious failures in the market for land for industrial and commercial development. We therefore believe it is likely that the scarcity of land is already reflected in the acquisition costs of landfill sites and hence scarcity is unlikely to represent an environmental or social externality.

Figure 1.1 illustrates the above discussion, i.e. the interaction of the scientific and economic studies.

When assessing the environmental impacts per unit of pollutant care has been taken to avoid double counting potential health impacts with the separate valuation of health effects. This is explained in more detail in subsequent sections of this report.

Figure 1.1 Interaction of scientific and economic studies on waste management



1.2 Classification of the literature

In this report we classify the literature into primary and secondary studies, although in practice there are overlaps in between. In this classification, primary studies estimate the economic value for a particular impact(s), where the estimates are the product of primary research. Secondary studies use (e.g. as part of an aggregation exercise in a particular policy context) the estimates obtained from the primary study in some form of benefit transfer exercise.

Figure 1.2 provides a summary of the state of the valuation literature with particular reference to waste management as of mid 2003. It has been compiled by reviewing the main secondary studies and then tracing these back to the underlying primary studies. It omits any primary research that is not covered in the known secondary studies. It also excludes some of the more recent work undertaken by Defra. These more recent studies include the current Air Quality Strategy Evaluation being conducted by Defra as well as two other ongoing studies, referred to in Section 6.

The key secondary studies are shown on the right in date order and the primary studies on the left. The studies underpinning the EcoSense model (part of the ExternE programme) are not shown because the information is not readily available. Although the EcoSense model is only shown feeding into ExternE, it is widely referred to and has been directly and indirectly used by several secondary studies. For reasons of presentation and time constraints, we have not shown all linkages in the evolution of the valuation literature.

1.3 Qualifications on valuation estimates

The external cost estimates provided per unit of pollutant must be seen in the context of large uncertainties. In the main they should be used as guides for orders of magnitude. Uncertainties are caused *inter alia* by:

- The compounding of uncertainties throughout the scientific and economic valuation methodologies. For example, there is uncertainty in estimates of release levels, the dispersion models, the dose response functions and the values for the final impact.

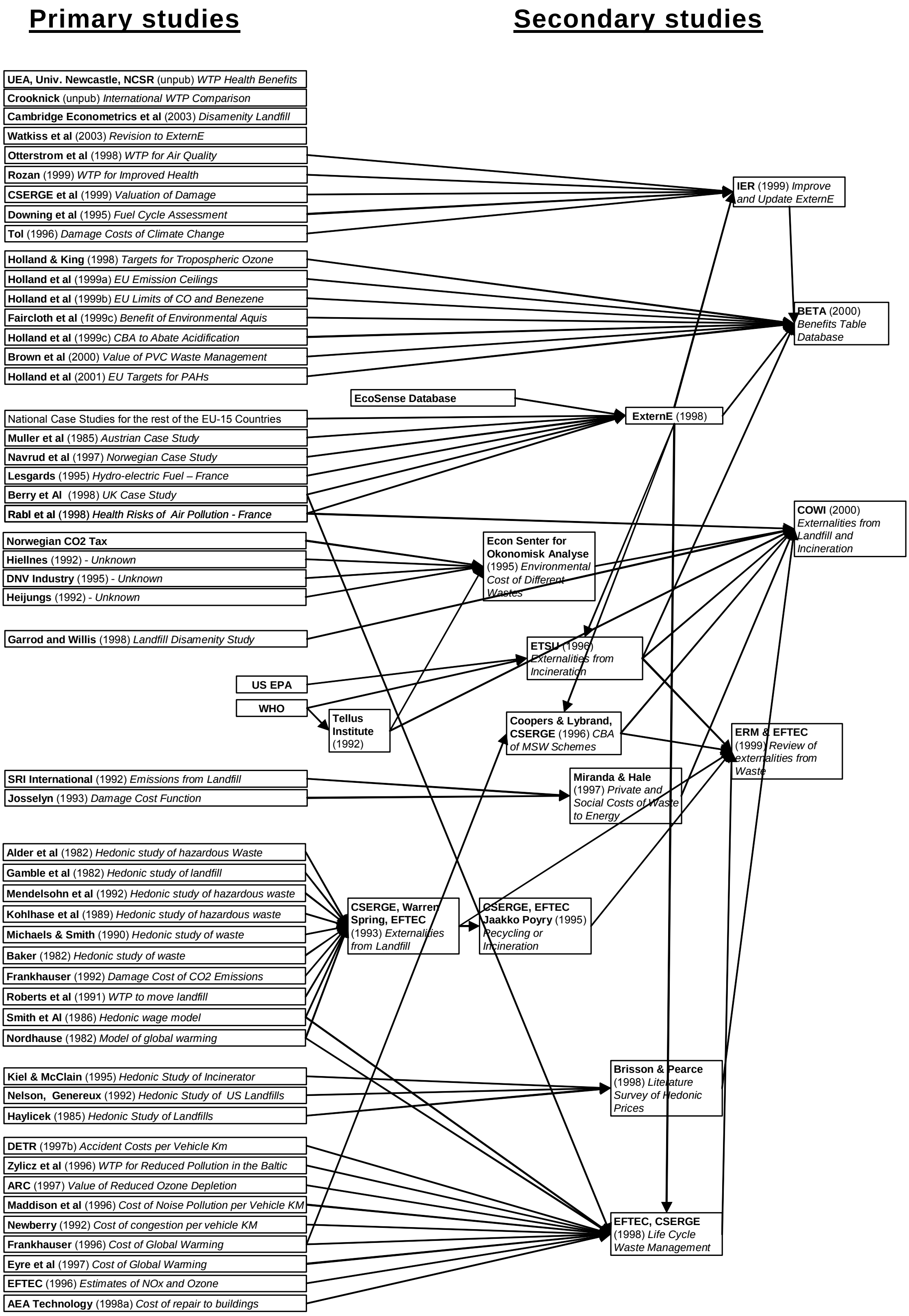
- Differences in methodologies used to calculate the values of impacts. For example some studies base their calculations on life cycle emissions whereas others examine the direct effects from disposal facilities alone.

- The lack of understanding of the chronic morbidity effects and "cocktail" effects of many pollutants.

- The site-specific nature of dose response functions which are influenced by local conditions, for example climatic patterns, spatial distribution of the population and socio-economic profile.

- A continuing (and currently inconclusive) debate on key economic value estimates, for example, the value of a statistical life and the damage costs associated with global warming.

Figure 1.2 Summary of the links between the valuation literature



2. Identification of releases

The purpose of this section is to identify the releases associated with each MSW management option analysed in the scientific study. Within this economic study we do not need to determine the level of any specific release, only that a release of a particular substance occurs in order to be able to place a value on its effects. These effects are included in the dose response functions. For health effects the dose response functions have been modelled in the scientific study. For environmental effects the dose response functions are implicit in the values of damage costs per pollutant obtained from the literature.

Tables 2.1 and 2.2 summarise the substances emitted to air, water and land from facilities that treat MSW and similar wastes that were investigated in the scientific study. Releases from facilities that do not treat MSW or similar wastes, for example hazardous waste, have not been researched in the scientific study and therefore are not covered in this economic study. These tables include releases from all the types of MSW treatment facilities included in the scientific study. The reader is directed to Section 2.2 of the scientific study for a complete description of releases from processes that treat MSW and similar wastes.

Table 2.1 Emissions to air From MSW treatment facilities

1,1-Dichloroethane
Ammonia
Carbon Dioxide
Chloride
Chlorobenzene
Chloroethane
Chloroethene
Dioxins and Furans
Hydrogen Chloride
Hydrogen Fluoride
Metals: arsenic, cadmium, mercury, nickel
Methane
Nitrogen Oxides
Particulate matter
Polychlorinated Biphenyls
Sulphur Dioxide
Tetrachloroethene
Total Volatile Organic Compounds

Source: Enviros (2004)

Table 2.2 Emissions to land / ground water / surface water from MSW treatment facilities

Aniline	Methyl tertiary butyl ether
Arsenic	Monohydric phenols
Biphenyl	Naphthalene
BOD	Nickel
Cadmium	Nitrates
Chloride	Nonyl phenol
Chromium	Organo-tin
COD	Pentachlorophenol
Copper	pH
Cyanide	Phenols
Di (2-ethyl hexyl) phthalate	Phosphorus
Dichloromethane	Polycyclic aromatic hydrocarbons
Dioxins and Furans	Suspended solids
Ethylbenzene	Thallium
Fluoride	Toluene
Lead	Xylene
Mercury	Zinc
Methyl chlorophenoxy acetic acid	

Source: Enviros (2004)

3. Review of secondary studies

This Section provides an overview of studies that have developed estimates of the external costs of pollutant releases. More complete descriptions of the studies referred to in this Section are provided in Appendix 1 along with reported values.

3.1 Overview of secondary studies

CSERGE, Warren Spring Laboratory and eftec, (1993), Externalities from Landfill and Incineration

The report calculates the externalities associated with the disposal of wastes to landfill and incineration per tonne of material. The emphasis of the study is on MSW. Whilst the study's primary focus is the UK, and most of the data used is based on UK studies, some information is transferred from US studies, particularly regarding disamenity due to the lack of relevant UK literature at that time.

CSERGE, eftec, Jaako Poyry, (1995), British Newsprint: Recycling or Incineration – An Evaluation of Alternatives

The study compared the impacts of disposing of newsprint via incineration and recycling. Emissions factors are provided in terms of kgs of pollutant per tonne of newsprint and as such they are of limited use in this study.

ExternE (1995 – 1998)

The ExternE project was conceived in the early 1990s under the EC Joule research programme with the aim of providing a consistent methodology for comparing the external costs of fuel cycles. It has since become the point of reference for much atmospheric emissions valuation work in Europe. The outputs are based on the ***EcoSense*** model developed by the Institute for Energy Research (IER) at the University of Stuttgart. The EcoSense model combines data on emission sources, stock-at-risk (e.g. population, material and crop inventories) with air dispersion modelling, dose-response functions and monetary valuations to derive damage costs for key energy related pollutants, including CO, SO_2, NO_x, NH_3, and PM. The monetary values used in the database are readily observable from publicly available information, as are aggregated costs per unit of pollutant for each Member State.

Coopers & Lybrand, CSERGE and eftec (1996), Cost Benefit Analysis of Different Municipal Solid Waste Management Systems: Objectives and Instruments for the Year 2000

The objective of the study was to project the total net economic cost of MSW treatment in each EU Member State, from 1993 to 2001. The total economic value was based on estimates of life-cycle emissions associated with each disposal route. Monetary values were derived from the available literature.

CSERGE and eftec, (1998), Life Cycle Research Programme for Waste Management: Damage Cost Estimation for Impact Assessment

The purpose of this report was to develop a database of environmental damage costs relating to waste management. The database was built up from various primary studies and covered six receptor categories: human health, buildings, crops, climate change, ecosystems and disamenity effects.

European Commission (2000), External Costs of Energy Conversion – Improvement of the ExternE Methodology and Assessment of Energy-related Transport Externalities

In 1999, the IER unit at Stuttgart University led a project to update and expand the EcoSense model to analyse transport related impacts.

ERM and eftec, (1999), Technical Review of Studies on the Externalities of Waste Management Options

The report examines eight studies on the environmental impacts of waste management. It concludes that many of the reviewed studies lack the coverage of emissions to provide a complete assessment of different waste disposal techniques and that not all the externalities can be valued. In particular, disamenity value was often overlooked.

COWI, (2000), A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration Waste

The report reviews seven studies to produce an estimate of the external cost of disposing of waste via incineration and landfill. The study concludes that whilst the methodology supporting the valuation of air pollution is becoming established, the pollution pathways and dose-response functions of pollutants released to soil and water are subject to greater uncertainties.

Netcen, (2002), BeTa: Benefits Table Database

The Benefits Table (BeTa) database, conducted by Netcen in 2002, is an update of the ExternE work. Using the same methodology of the ExternE database, the BeTa work addresses the impacts of four gases (SO_2, NO_x, VOCs and PM) from three specific sources of emissions: all sources of emissions in rural locations, ground level emissions in urban areas (primarily traffic), and emissions from shipping.

3.2 Summary of secondary study estimates

The rest of this section provides a simple summary of the average external values obtained from the studies identified above. Results have been inflated and converted into £2003 in both tables using the procedure described in Appendix 4.

The purpose of these tables is to highlight the orders of magnitude and range of possible damage costs for the substances shown. The large ranges in values for the same pollutant are caused in part by the state of development of the valuation and benefits transfer literature at the time of the study, different study methodologies, the types of facilities studied and the country of focus.

It is important to highlight that there are very large uncertainties in the figures presented in these tables, and that they should not be used for benefits transfer without a more detailed appreciation of the assumptions behind the figures. It is beyond the scope of this project to describe each secondary report in detail, but the main ones used to compile Tables 3.1 and 3.2 are summarised in Appendices 1 and 2.

The most useful application of these numbers is in carrying out sensitivity analysis on the core results derived using the values obtained from the primary studies (see Section 4). If the conclusions drawn from the combination of the scientific and economic studies on the value of the relative environmental and health impacts of alternative waste management options are sensitive to the unit values in these secondary studies then further examination of the data sources and assumptions should be undertaken.

In Tables 3.1 and 3.2 where the value is derived from more than one study, we have taken the arithmetic average of all relevant studies. Whilst we have omitted some study results because of extreme outlying values, no weighting has been applied to take account of the quality of each study's results. Moreover these tables do not differentiate between types of studies or methodologies. There will be significant differences in values for the same pollutant between point source and line source emissions and between rural and urban environments. Unit damage costs will be much higher in the latter case because emissions disperse less and the stock-at-risk is larger.

Table 3.1 shows the summary for the key air emissions of SO_2, NO_x, PM (2.5 and 10), VOCs, CO and CO_2. These six substances are highlighted because of their importance as general air pollutants. There is also a bias in the economic valuation literature towards these pollutants because of the focus of the research on fuel cycles. Although waste treatment processes emit a more varied range of pollutants (as indicated by Tables 2.1 and 2.2) and in different quantities to fuel conversion processes, previous research has suggested that the total external costs from some waste management processes, notably incineration, are dominated by these key pollutants.[4]

Table 3.2 shows the values for a wide range of other pollutants. These contain the organic and metallic compounds that are known to be emitted from waste treatment processes. As noted above, much of the scientific analysis on the effects of air pollution has centred on the above six key pollutants. Less is known about the effects of these other pollutants on the human population and general environment.

4 AEA Technology, 1996

Hence the range of uncertainty for these pollutants is wider than for the six key pollutants.

In both these tables, we have categorised the value by type of impact. Five main categories are used, referring to impacts on:

- Health (mortality and morbidity)
- Agriculture (including forestry)
- Buildings
- Eco-system
- Climate Change

Because several studies are used for each pollutant, means and ranges are shown. The reference sources of the underlying studies are not shown here for presentation purposes, but are provided in Appendix 2.

We have not identified the extent to which health effects shown in these secondary studies include chronic as well as acute effects. This will depend on the nature of the underlying primary research from which the values in the secondary studies are drawn.

On both tables a blank cell means that no value for that effect is given in the literature. In Table 3.1, estimates for health impacts in some cases include values associated with other effects, although by far the major contribution to the value is from health effects for all pollutants with the exception of CO_2.

In Table 3.2 “included” indicates that a value for that effect is acknowledged in the literature, but it is incorporated in the health effects value. Also this table does not present the results from ECON (1995) as they are three orders of magnitude higher than other studies. [5] This is denoted in the table by “no useful data”.

Table 3.1 Summary of Values for Key Pollutants from Secondary Studies (£/tonne pollutant, 2003 prices)

Sub-stance	Number of studies	Mean / range	Mortality & Morbidity	Agricul-ture	Buildings	Eco-system	Climate Change	Total
CO_2	9	Mean					10	10
		Range					1 - 38	12
CO	2	Mean						2
		Range						0
NOx	10	Mean	7,519	-135 [(a)]	205			7,550
		range	1,500 – 29,000	-135	140-269			1,500 – 29,000
PM2.5	3	Mean	105,600					105,600
		range	2,000 – 300,000					2,000 – 300,000
PM10	10	Mean	34,900		200			35,000
		range	1,800 - 226,500		0			1,800 – 226,700
SO_2	11	Mean	6,990	14	578	8		7,590
		range	1,700 – 15,000	10 - 18	370 -962	1 – 13		1,700 – 16,000
VOC	4	Mean	1,000					1,000
		range	500 – 1,500					500 – 1,000

Note: [(a)] The negative value indicates a yield enhancement effect – i.e. an external benefit.

5 ECON, (1995) Senter for Okonomisk analyse, Environmental Costs of Different Types of Waste.

Table 3.2 Summary of values for other pollutants from secondary studies(£/kg pollutant, 2003 prices)

Substance		Mortality & Morbidity	Agri-culture	Buildings	Eco-system	Climate Change	Total
1,2,3-Trichloro-propane		6,900					6,900
1,3 Butadiene		70					70
2-Butanone		820					820
4-methyl 2-Pantanone		820					820
Acetone		410					410
Antimony (Sb)		102					102
Arsenic (As) (5 studies)	Mean range	400 1.2 – 1,570					400 1.2 – 1,570
Barium (Ba)		No useful data					
Benzene		1.8					1.8
Beryllium (Be)		38,000					38,000
Bisphthalate		240					240
Cadmium (Cd) (5 studies)	Mean range	201 61 – 700					201 61 – 700
CFC 11		7.5					7.5
CFC 12		6.2					6.2
CFC 13		6.8					6.8
Chromium (Cr) (3 studies)	Mean range	1,220 99 – 2,320					1,220 99 – 2,320
Copper (Cu)		42					42
Diethypalate		240					240
Dioxins (TEQ) (2 studies)	Mean Range	7,400,000 1,600,000 – 13,000,000					7,400,000 1,600,000 – 13,000,000
Halon 1211		38			Included		38
Halon 1311		90			Included		90

Substance		Mortality & Morbidity	Agri-culture	Buildings	Eco-system	Climate Change	Total
Hydrogren Chloride (HCl)		No useful data					
Hydrogen Fluoride (HF)		No useful data					
Lead (Pb)		1,220					1,220
Mercury (Hg)		No useful data					
Methane (CH_4) (5 studies)	mean Range					0.5 0.006 – 1.8	0.5 0.006 – 1.8
Methyl Chloride		No useful data					
N_2O (2 studies)	Mean Range					3,8 2.1 – 5.4	3,8 2.1 – 5.4
Nickel (Ni) (5 studies)	Mean Range	58 2 – 230					58 2 – 230
Nitrates		(a)			9.7		9.7
p-Cresol		No useful data					
Phenol		No useful data					
Selenium (Se)		No useful data					
Tin		No useful data					
Toluene		No useful data					
Trans-1, 2-dichloroethyl ene		No useful data					
Vanadium		No useful data					
Vinyl Chloride (VC)		No useful data					
Zinc (Zn)		No useful data					

Notes: (a) Although nitrates do have a direct mortality effect as an air pollutant (as a secondary pollutant formed from NO_x emissions) a suitable valuation estimate was included in the literature sources covered in this review.

4. Review of primary valuation studies

This section of the report focuses on the *primary* economic valuation studies that have estimated economic values for the range of impacts on human health and the environment that are typical of waste management options. Primary economic valuation studies employ various techniques to estimate the economic costs and benefits that result in, for example, a change in environmental quality. Following standard economic theory, economic costs and benefits are measured through people's preferences, specifically how they trade-off various goods with money.

These 'economic valuation' techniques fit into two categories: techniques which reveal people's preferences through the market (e.g. hedonic price techniques, by which for example housing prices are analysed to reveal the implicit (costs) benefits of local (dis)amenities) and stated preference techniques where surveys are used to create hypothetical markets whereby people state their willingness to pay (or willingness to accept) to achieve (or forgo) an environmental change. Both these types of techniques have been employed in the literature reviewed here.

An examination of these original economic studies sheds light on what lies behind the aggregated values for the economic cost per tonne of waste presented in the secondary studies. The secondary studies have employed different estimates from the primary economic valuation literature for different impacts, as new studies have become available, and have summed these together to arrive at total economic cost across all impacts. The purpose of this section of the report is to examine (a) the range of primary study estimates employed in the past for these impacts and (b) new studies that have emerged since, in order to select the best available estimates for individual environmental and health impacts. The intention is that these data will inform the Government in its efforts to build a new, up-to-date estimate of the external costs per tonne of waste based on the impacts analysis from the scientific study. This impact list provides the basis for the literature search.

The application of results from economic valuation studies to a new context is called 'benefits transfer'. While the name suggests the transfer of economic benefit estimates, the same technique applies to the transfer of economic cost estimates. When transferring from the original study context to a new context, there is always a certain level of uncertainty involved which will reduce confidence in the final result. This uncertainty results from transferring values from:

- Another country: i.e. the results of a study on the disamenity costs of an incinerator in the US, when applied to the UK, will suffer from errors due to cultural, geographical and social differences between the two countries which imply the public's preferences for these impacts in two countries could be different.

- Studies that are not statistically robust: oftentimes the only available study for a certain impact may suffer from statistical problems that come from having a small sample size (in the case of stated preference survey techniques) or other data and design related issues.

- Studies examining a different context or change: the study may have examined a slightly different context or environmental change. For example, willingness to pay to avoid a respiratory hospital admission when applied to a cardiovascular hospital admission could result in an unknown error.

- Studies that are more than about 10 years old: economic valuation techniques are continually being improved and thus less confidence should be applied to older studies.

In selecting recommended values, these points have been the key selection criteria. Therefore, preference has been given to UK studies, with robust results that have been conducted in recent years and which examine a closely matched context to that of this study. Preference is always given to meta-analyses which have examined a number of similar studies and provided recommended estimates from this analysis. Fortunately, there are a number of studies that meet most of these criteria. However, for some impact types, studies that are good matches are thinner on the ground, and thus these studies have either been (a) presented and not recommended or (b) recommended with 'health warnings'. While the results may suffer from error, they will provide an indication of the magnitude of cost associated with that impact.

It is worthwhile reiterating here that, with regard to air pollution, this section also presents economic cost estimates per tonne of emission from secondary studies. However, for the most part, these estimates are based on the aggregation of the most up-to-date primary study estimates in the UK context. The reason for presenting these 'secondary' estimates in a section about primary studies relates to the information provided by the scientific study and the potential usefulness of employing both emission related costs and impact related costs when deriving the cost estimates per tonne of waste.

In this section we also address the value of displaced energy use. Appendix 4 contains details of currency and year conversions applied to the data presented in the report.

4.1 Health effects

The health impacts of waste management options identified in the scientific study are:

- Deaths brought forward (acute mortality) – that is, deaths occurring sooner than would otherwise occur.[6] This effect is observed following exposure of elderly and/or sick people to elevated levels of some air pollutants. In the scientific study, deaths brought forward do not include deaths due to cancer caused by airborne carcinogens.

- Respiratory hospital admissions

- Cardiovascular hospital admissions

- Cancers[7]

- Birth defects
  - Neural tube defects
  - Cardiovascular defects
  - Hypospadias and epispadias
  - Abdominal wall defects
  - Gastroschisis and exomphalos
  - Low birth weight
  - Very low birth weight

Due to the limited scope of the dose response functions used in the scientific study (which are based on observed effects), chronic mortality or morbidity are not identified. This has the potential to underestimate the total health impacts of the pollution from MSW management. Thus, this section presents a broader range of economic valuation work on chronic mortality and morbidity to provide a more complete picture of the health impacts of pollutants and releases from MSW management. Table 4.1 summarises the health effects covered by the scientific study and the economic literature.

6 The scientific study employs exposure-response coefficients and subsequent advice on their application from the Department of Health's Committee on the Medical Effects of Air Pollutants report (COMEAP, 1998, 2000 and 2001).

7 The scientific study employs World Health Organisation's unit risks of developing cancer when exposed to the concentration of a pollutant.

Table 4.1 Health effects covered by scientific study and economic literature

	Scientific Study	Economic Literature
Mortality		
Deaths brought forward	√	√
Lost life years		√
Respiratory illnesses		
Hospital admissions (all types)	√	√
Casualty / emergency admissions		√
Lower respiratory symptoms		√
Additional cases of cancer		
Lung	√	√
Leukaemia	√	√
Haemangiosarcoma	√	
Fatal (all types)		√
Non fatal (all types)		√
Cardiovascular hospital admissions	√	
Birth defects		
Neural tube defects	√	√
Cardiovascular defects	√	√
Hypospadias and epispadias	√	
Abdominal wall defects	√	√
Gastroschisis and exomphalos	√	√
Low birth weight	√	
Very low birth weight	√	

4.1.1 Deaths brought forward

The scientific study provides estimates of deaths brought forward and defines this as ‘deaths occurring sooner than would otherwise occur from other causes…this effect is observed following exposure of elderly and sick people to elevated levels of some air pollutants’. This definition is consistent with episodes of ‘acute mortality’, whereby a sharp increase in pollution levels results in an increase in the death rate. This is different to ‘chronic mortality’, in which people are exposed to pollution over long periods and their lives are foreshortened accordingly. In both cases, the effect is a foreshortening of lives of the elderly members of the population.[8] Chronic mortality is more likely to be the result of pollution from landfill and incinerators, although this wasn’t examined by the scientific study.

The estimates from the scientific study of deaths brought forward are extremely small. The study estimates that one death might be expected to be brought forward due to emissions from an individual waste management facility every 100 to 1,000 years, depending on the

8 Although infants and children can also be the victims of acute and chronic mortality from air pollution, insufficient research on the value of statistical life for very young people precludes consideration of this effect in the analysis.

type of facility. This is effectively a negligible effect, and the review of the literature that follows should be viewed in light of this result.

The literature on valuing deaths is vast and it is beyond the scope of this study to provide a review and analysis of all the literature and issues (for an in-depth review see, for example, Pearce, 2000). Here, the discussion introduces the various approaches to valuing deaths and presents the figures currently being used in policy-making, and then recommends an appropriate figure.

The majority of the economic literature on this topic focuses on estimating the value of statistical life (VOSL) through analysis of markets where risks are traded for money - the labour market being the most studied of these. See Box 4.1 for an overview of how VOSL is calculated. The most recent meta-analysis of this literature is provided by Viscusi and Aldy (2002), who conclude that the median value for prime-age workers is $7million in the United States. The authors suggest that this value may be higher in the UK, where much larger risk premiums are found in the labour market.

Drawing from the same body of literature, the European Commission's DG Environment employs a more conservative estimate of VOSL of €1million (with a range of €0.65 to €2.5million) for use in DG Environment cost benefit analyses.

The approach employed by the UK Department for Transport (the then DETR, 1999) to value casualties from road accidents is to calculate the sum of loss of output (i.e. the present value of the expected loss of earnings plus any non-wage payments paid by the employer), the ambulance costs, the costs of hospital treatment, and the human costs, based on VOSL values. This results in a value of lost life of £1.25million (£,June 2002).

However, there is growing consensus that applying VOSL estimates to acute mortality from air pollution is inappropriate. Acute episodes tend to shorten life by weeks or months in an elderly and/or already sick population, whereas VOSL estimates are generally derived from studies that examine risk of death to people in the prime of life, where the implicit loss of life years is much greater. The valuation of life years lost (VLYL) approach is an attempt to overcome this problem. So far this has involved simply adjusting VOSL figures for factors such as age and quality of life (see Box 4.1). While economic theory would suggest that WTP will decline with age, there is no quantitative evidence for this, neither is there evidence for how health status affects WTP (Cropper, 2000).

An exercise in adjusting VOSL for air pollution related mortality was undertaken in the Department of Health's Economic Appraisal of Health Effects of Air Pollution (EAHEAP, 1999). Starting with a VOSL of £2million, EAHEAP concluded that the combined effect of age and increasing physical risk aversion might plausibly result in lower VOSL for people over 65. Using a factor of 70% this resulted in an upper-bound value of £1.6million. Adjusting for health-related quality of life,

life expectancy and updating for inflation, an adjusted range of **£**3,100 to £110,000 was recommended for respiratory deaths. The total range resulting from these adjustments is reported as £3,100 to £1.6million. The main shortcoming of the EAHEAP recommendations is that the VOSL estimates, although adjusted, are not based on primary willingness to pay studies that focus on risks associated with exposure to air pollutants, as these studies are lacking from the literature. Also lacking is substantive empirical evidence about how WTP for mortality risk reductions is influenced by age, income and health status. As Rabl (2003) suggests the appropriate measure in air pollution contexts should be years of life lost, not VOSL. Defra has commissioned a WTP study to address some of these gaps in this literature, the results of which are forthcoming. That study should provide additional evidence for the recommendation of an economic cost for mortality in the context of air pollution (see Section 6 for more details).

For the purposes of this study, currently the best available estimate that takes into account the air pollution context is the low to medium range from EAHEAP of **£3,100 - £110,000** for a death brought forward in the context of air pollution. Employing the high value of £1.6million is likely to be a significant overestimate for the reasons given above.

Box 4.1: Value of Statistical Life (VOSL) vs Value of Life Year Lost (VLYL)

The approach to calculating VOSL is based on the estimation of the willingness to pay (WTP) for a small reduction in the risk or probability of death, where people are asked to ‘buy’ or ‘sell’ small variations of the probabilities associated with death by various causes. The aggregation of individual values over a whole population leads to the VOSL (or the value of a statistical death avoided). This is summarised by the following equation:

$$VOSL = \sum_i WTP_i \cdot \Delta r_i / \sum_i \Delta r_i$$

where ΣWTP_i = sum of individual WTP for the change in risk over N individuals;

Δr_i=the change in risk; N=number of persons exposed to the risk, and $\Sigma\Delta r_i$=the number of statistical lives gained or lost.

There are two approaches to valuing the value of a life year lost (VLYL):

- by calculation of the implicit WTP for the loss of each year of remaining life, using VOSL estimates, the age profile (and hence probability of survival) of the sample population and a discount rate.

- through specially designed WTP studies, where what is valued is an extension to expected lifetime rather than the reduction of a risk of not achieving the expected lifetime.

4.1.2 Respiratory hospital admissions

The economic cost of any illness episode is composed of the following parts: the value of the time lost because of the episode (i.e. productivity loss), the value of lost utility because of the pain and suffering (i.e. WTP to avoid an episode) and the costs of treating the episode (i.e. health service costs) - also known as the Cost of Illness (COI). The most up-to-date valuations for all of these components of the cost of respiratory admissions are found in CSERGE et al (1999).

The CSERGE et al (1999) report contains a number of research papers on benefits transfer and environmental damage in the EU, including a contingent valuation study (Day et al, 1999) which surveyed samples in five European countries to estimate WTP to avoid episodes of ill-health. Day et al (1999) is the first primary valuation study on morbidity to take place in the EU. All previous estimates were borrowed from US studies. In another paper in the report, Dubourg (1999) presents costs of illness estimates for the same five countries. These papers are summarised in Appendix 3.

The studies differentiate between the cost of a hospital admission and a casualty admission. Each is described as follows:

- Respiratory Hospital Admission (RHA): hospital admission for treatment of respiratory distress where symptoms include persistent phlegmy cough with occasional coughing fits, gasping breath, fever, headache and tiredness. It is assumed that the patient stays in hospital receiving treatment for three days followed by five days at home in bed.

- Respiratory Hospital Casualty Admission (RHCA): visit to hospital casualty for oxygen and medicines to assist breathing problems caused by respiratory distress where symptoms include a persistent phlegmy cough with occasional coughing fits, gasping breathing even when at rest, fever, headache and tiredness. Patient spends four hours in casualty followed by five days at home in bed.

Table 4.2 summarises the results of the CSERGE work for the UK in terms of RHA and RHCA, and presents a total economic damage cost for each. The estimates have been adjusted to £2003 prices in the final column (see Appendix 4 for details of the conversion).

Table 4.2 UK estimates for costs of respiratory hospital admissions

		Reported values	£,2003
WTP to avoid Respiratory Hospital Admission (per ill health episode)	A	164 (£1997)	177
WTP to avoid Respiratory Casualty Hospital Admission	B	131 (£1997)	141
Per diem productivity costs from absenteeism	C	53.35 (€ 1997)	59
Cost of hospitalisation (per inpatient day)	D	187 (£1996)	205
Cost of emergency room visit (per visit)	E	52.24 (£1997)	56
TOTAL (Hospitalisation) = A + (C x 8days) + (D x 3). Per episode.			**1,264**
TOTAL (Casualty/ER) = B + (C x 5days) + E. Per episode.			**492**

Source: CSERGE et al (1999)

4.1.3 Cardiovascular Hospital Admissions

There is no specific economic valuation literature that investigates WTP to avoid a cardiovascular hospital admission. It is expected that WTP to avoid this type of hospital admission would be higher than for respiratory admissions because the implications tend to be more severe and the suffering felt over a longer period of time. When it comes to the costs to the health service, Netcen (2002) assumes a stay in hospital of seven days. This number of days is also within the range suggested by the EAHEAP report. EAHEAP estimated that the average length of stay of patients admitted to hospital due to cardiovascular disease was between 6.4 and 10.6 days (the length of stay is dependent on the patients age)[9]. Although it is not documented in the literature, we assume that the days of absenteeism from work for a cardiovascular hospital admission would be proportionate to those taken after an RHA (i.e. eight days). Thus, a lower bound estimate for a cardiovascular hospital admission can be provided by the WTP to avoid an RHA, plus the productivity losses from time off work, plus the cost of hospitalisation (assuming a seven day stay), which gives **£2,674** per episode.

4.1.4 Cancers

Cancer risks are reported in the scientific study for lung cancer, leukaemia and haemangiosarcoma. The economic literature has been reviewed in relation to these specific cancers.

In the scientific study there is no separate analysis to differentiate between the number of these cancer cases that result in a fatality.

9 Data in the EAHEAP report is taken from the NHS Hospital Episodes Statistics 1994/95.

Although it might be possible to calculate (based on health statistics) the percentage of cases of these cancers that lead to mortality, such an analysis is outside the scope of this study. Instead, the approach taken here is to present valuations for both fatal and non-fatal cancers, which will provide upper and lower bound estimates for the costs of cancer cases.

Non Fatal Cancers

As with other illnesses, the approach to valuing the economic cost of non-fatal cancers should ideally include WTP to avoid suffering from the illness as well as the cost to the health service and losses in productivity. The economic valuation literature on non-fatal cancers (NFCs) is summarised in Table 4.3 and shows a mix of approaches, but none of the figures include the sum of WTP, COI and uncompensated/uninsured forgone earnings, and thus are incomplete estimates of economic cost.

Table 4.3 Economic valuation of non fatal cancers (£, 2003)

Author	Country	Method	Value per case €1999	Value per case £2003	Type of cancer
Rowe et al.,1995	USA	COI	186,000	199,000	NFCs generally
ExternE	Europe	COI + forgone earnings (adjusted for WTP/COI)	450,000	482,000	NFCs generally
Aimola, 1998	Italy (Sicily)	WTP (using CVM)	50,000	53,500	Lung cancer
			730,000	781,000	Leukaemia

Source: adapted from Pearce (2000)

Rowe et al. (1995) adopt a value based on the US costs of treating cancers ('cost of illness', COI) and then multiply this by 1.5 to arrive at WTP on the basis that, where COI and WTP studies are available, WTP appears to be 1.5 times the COI. This procedure is not considered satisfactory, as there are few studies that estimate both COI and WTP. Moreover, the Rowe et al. COI value dates from the mid-1970s (Pearce, 2000).

ExternE uses a figure of €450,000 for an NFC (or £482,000 in 2003 prices), which is derived as follows. In earlier ExternE work the authors started with a value of a non-fatal cancer of €250,000 that was based on the direct costs of illness plus forgone earnings. Adjusting for inflation, this becomes €300,000. The authors then apply the WTP/COI ratio recommended by Rowe et al (1995) and on this basis arrive at a value for non fatal cancers of €450,000. This approach is not recommended for the same reasons that Rowe et al's is not.

Aimola (1998) uses the contingent valuation method to elicit cancer risk valuations from a small sample of the population in Sicily. The cancers included leukaemia and lung cancers.

Taking this literature into account, one could use the ExternE COI and forgone earnings estimate (€300,000) and add to this Aimola (1998) WTP estimates for lung cancer and leukaemia. This would give a total economic cost estimate for lung cancer of £375,000 and leukaemia of £1,100,000 (£,2003). While these are the best estimates available for non-fatal cases of leukaemia and lung cancer, the applicability of a Sicilian study to the UK is low. Better estimates are available for fatal cancers as reported below.

Fatal cancers

Fatal cancer cases could arguably be valued at the same cost as VOSL, although it is usually assumed that people would pay more to reduce the risk of death from cancer (as opposed to other causes of death) because of the morbidity that usually precedes fatal cancer (Cropper, 2000).

One approach is to add a ‘cancer premium’ to the VOSL estimate that is equivalent to 50% of that value. Thus, if €1million was the standard VOSL, then mortality due to cancer would be valued at €1.5million, where the extra €0.5million is attributable to the period of ill health before death.

In line with the above approach the UK study on Surface Transport Costs and Charges (Sansom et al, 2001) used a value of £1.6million per fatal cancer case, based on discussion with the Department of Health (DoH). This is the medium estimate of a range from £1.1million to £2.2million and is equivalent to £1.15m, £1.66m and £2.26m in 2003 prices.

One approach, used by the EU and outlined in the BeTa study for valuing chronic mortality (Netcen, 2002) is to discount the VOSL figure (the EU employs a value of €1million) by 4%, recognising that the effect of an emission now will be spread out over a number of years, giving a value of €490,000 per case of chronic mortality. Note that this is the opposite approach to valuing cancer mortality outlined above, where the cost of a fatal cancer is greater than VOSL.

Finally, a third approach is to estimate the value of a life year lost (VLYL) and multiply this by the mean years of life lost (YOLL) for different types of cancer. Tables 4.4 and 4.5 below from Markandya (1997) present the results of this approach. Table 4.4 presents estimates of VLYL, based on different discount rates. In the table, the choice of VLYL depends on the latency period of the cancer. Table 4.5 presents estimates of the latency period and of YOLL for leukaemia and lung cancer. These are then used to calculate the aggregate value of a fatal case of these cancers.

The choice of discount rate affects the results. If a 3% discount rate is chosen (the closest rate to 3.5% recommended by UK Government for public sector appraisals (HM Treasury, 2003)), the value of a fatal case of lung cancer and leukaemia are €1,810,000 and €1,080,000, respectively.

Table 4.4 VLYL for different latencies and discount rates (1995,ECU)

	Discount rate		
	0%	**3%**	**10%**
VLYL	98,000	155,000	312,000
Latency 0 years			
female	98,000	122,123	169,198
male	98,000	120,187	162,059
Latency 30 years			
female	98,000	53,820	11,175
male	98,000	53,810	11,133
Latency and risk distributed over 30 yrs			
female	98,000	84,680	61,269
male	98,000	83,969	59,371

Source: Markandya (1997)

Table 4.5 VLYL for different cancers

	Leukaemia		**Lung cancer**	
Latency (l) in years	8		15	
Estimated mean YOLL	22		16	
Discount Rate	**(1995, ECU)**	**(£,2003)**	**(1995, ECU)**	**(£,2003)**
0% (VLYL= 98,000)	2,160,000	2,315,000	1,570,000	1,682,000
3% (VLYL=155,000)	1,810,000	1,940,000	1,080,000	1,160,000
10% (VLYL=312,000)	1,180,000	1,265,000	418,000	448,000

Source: Markandya (1997)

The cost of illness (COI) should be added to the values presented in Table 4.5 (where COI = £450,000, an ExternE estimate). If the estimates calculated using a 3% discount rate are taken from the Table, and the sum of these estimates and the COI are converted to £,2003, this gives a total value per case of **lung cancer** of **£1,480,000** (£,2003) and for **leukaemia** of **£2,260,000** (£,2003). These results are consistent with the figures recommended by the Department of Health in the Sansom et al study (2001).

Clearly, the approaches presented here are very different. Adding a cancer premium to the VOSL used to value other mortalities (such as road accidents) implies that it is better to die quickly than suffer longer before death. Using the VLYL approach provides greater distinction between different types of cancer and each type's impact on suffering and longevity, but is potentially flawed (Rabl, 2003). The ranges from DoH and the Markandya analysis are very close and thus point to a recommended range which combines the two of **£1.15 - £2.26million** (£,2003) in addition to the specific values provided for each cancer type. This is the range recommended for use from this study for general cancers.

4.1.5 Birth defects

To value the costs of birth defects we use a US study conducted by Waitzman et al (1995), which adopts a COI approach to value the economic costs of birth defects and cerebral palsy. Estimates were made of the direct costs of medical, developmental and special education services and the indirect costs of lost work and household productivity attributable to premature morbidity and mortality of the cohort of persons born in California in 1988 with any of 17 of the most clinically important birth defects and cerebral palsy. Intangible costs, 'pain and suffering', are excluded from this approach. Costs for a number of the birth defects identified in the scientific study are presented in Table 4.6.

Table 4.6 Economic cost of birth defects in the USA

Birth defect type	Cost per new case	
	$,1992	£,2003
Neural tube defects	294,000 – 503,000	246,000 - 422,000
Cardiovascular defects	267,000 – 505,000	224,000- 423,000
Hypospadias and epispadias	NV	-
Abdominal wall defects	176,000	148,000
Gastroschisis and exomphalos	108,000	90,500
Low birth weight	NV	-
Very low birth weight	NV	-

Source: Waitzman et al (1995)

NV: no value reported in the literature.

4.1.6 Summary values of health impacts

Table 4.7 provides a summary of the estimates presented in this section for the valuation of health impacts arising from MSW, for which impacts are quantified in the scientific study, i.e. deaths brought forward, respiratory admissions, cases of cancer and birth defects.

Table 4.7 Summary Values of Health Impacts from Primary Studies

Impact from Scientific Study	Primary Economic Study Reference	Change being valued	Values per case (£,2003)
Deaths brought forward	various	Value of Statistical Life	3,100 – 1,600,000 [b]
Respiratory admissions (PM10, SO_2 NO_x)	CSERGE et al (1999)	Casualty/Emergency Room Admission Hospital admission	492 1,264
Cardiovascular hospital admission	Inferred from CSERGE et al (1999)	Hospital admission	2,674
Non Fatal Cancers [a]	Own calculations based on Pearce (2000) [a]	Leukaemia and lung cancer [a]	53,000 – 780,000 [a]
Fatal Cancers	Own calculations based on Markandya (1997) + COI + forgone earnings	Leukaemia Lung cancer	2,260,000 1,480,000
	DoH	All cancers	1,150,00 - 2,260,000
Birth defects (around landfill)	Waitzman et al (1995)	Neural tube defects	246,000-422,000
		Cardiovascular defects	224,000 – 423,000
		Hypospadias and epispadias	No value
		Abdominal wall defects	148,000
		Gastroschisis and exomphalos	90,500
		Low birth weight	No value
		Very low birth weight	No value

Notes: [a] Values are not recommended for benefit transfer to UK MSW facilities

[b] Although the full range of £3,100 to £1.6m is shown in this table, the lower range of £3,100 to £110,000 is used as a recommended estimate in Table 5.1. See Section 4.1 for a discussion of the reasoning.

4.2 Environmental effects

The scientific study does not provide data on the environmental effects of air pollutants in a form that is consistent the application of cost estimates of individual effects through benefits transfer. Hence the analysis in this section has to rely on studies that have already estimated the environmental impacts of air pollutants, using dispersion modelling and dose-response functions, and have reported the damage costs for these emissions within £/tonne of pollutant. The results of these studies are shown in Section 4.4. First though we review the valuation literature on specific environmental effects.

4.2.1 Emissions to air

The receptors of the environmental impacts of air pollution are generally categorised as follows:

- Buildings and man-made materials
- Climate Change (which will include a large element of health damages)
- Agriculture (including forestry)
- Eco-system (all flora and fauna, including forestry)

One widely used source of data on valuing environmental damages related to air pollution is the European Commission study, ExternE, on the external impacts of energy use. More recent attempts at valuing damages from air pollution are revisions of the impact-pathway approach adopted by ExternE using more up-to-date dose-response data and/or economic value estimates. Air pollution damages are generally presented in terms of £ per tonne (or kg) of pollutant and include, along with environmental damages, the value of health effects.

Damage to Buildings and Materials

Functional buildings and materials

Studies examining the impact of acidic depositions on materials are generally restricted to the most significant cause of damage, sulphur dioxide (SO_2). There is also a smaller literature on the impact of ozone (from VOC and NO_x production) on building and materials (Lee et al 1996; Holland, 1998). Different materials are examined such as rubber goods (for ozone damage) but, in general, the literature covers ‘primarily functional buildings’, such as residential, commercial and infrastructural. Little is known about the economic value of air pollution damage to cultural assets (see below).

Most approaches estimate the economic damage to buildings and materials from the cost of replacing the lost material, i.e. repair costs. There are three components to estimating the economic cost of damage to buildings and materials from acidification, using this approach:

- Determining suitable dose-response functions;

- Definition and measurement of the stock at risk. In the UK this information is derived from building survey data of typical construction materials per building type along with number of each type of building; and

- Estimation of the unit repair or replacement cost (£ per m^2 of degraded surface area). The approach assumes that maintenance is carried out after a given thickness of material has been lost.

Using repair/replacement costs will overestimate the amount of replacement activity that occurs and underestimate the true costs of air pollution on buildings to society (Calthrop et al 1997). Society's willingness to pay to avoid the damage should also be a component of the total economic value of the damage. Overall the bias in using replacement costs is assumed downwards. Thus any unit damage values derived from such studies should be interpreted as a lower bound of economic costs.

Sulphur Dioxide

Holland and Krewitt (1997) report estimates of materials damage from SO_2 for wider Europe as the outcome of a postulated acidification control strategy in the European Union. It is possible to take the estimates of monetary benefits in the year 2010 and divide them by the reductions in emissions of SO_2 in that year in order to derive a unit damage value (or benefit of reduction) for SO_2. Clearly, the result is very approximate and cannot be applied to any specific source of emission. The number is, in effect, a Europe-wide average. Average benefits are seen to fall as the control strategy gets stricter as would be expected. Thus different unit damage values are recommended for different SO_2 reduction policies. For example, the benefits of the maximum feasible reduction strategy are reported as €1,661million for a 12.02million tonne reduction in emissions of SO_2. This gives a benefit or avoided damage cost of €138/tonne SO_2. The benefits of an intermediate scenario are also reported as €880million for a 3.38million tonne decline in emissions of SO_2. This gives a benefit or avoided damage cost of €260/tonne.

Thus the analysis of a very strict SO_2 reduction policy would make use of €138/tonne SO_2, but for any intermediate control strategy the relevant figure is around €260 /tonne.

AEA Technology (1998) investigated the damage to galvanized steel and zinc, silicate paint and natural stone on modern buildings caused by concentrations of SO_2 and ozone. They use repair costs rather than WTP to avoid damage estimates for the valuation of damages in monetary terms. Thus the results are interpreted as lower bound estimates of true economic costs.

The study reports benefits of some €720million for a 12million tonne reduction in SO_2 emissions under a maximum feasible scenario, implying an average benefit of only €60/tonne SO_2. No estimate is available for intermediate scenarios but on the basis that other estimates suggest average damages of about double the maximum feasible reduction, €120/tonne SO_2 is recommended. The study does not report ozone reduction levels, thus, it is not possible to determine a unit damage value for ozone impacts to materials from this study.

Cultural heritage

Recent work on air pollution impacts does not include damages to cultural heritage, such as cathedrals and other fine buildings and statues. The reason for this is the assumption that as urban SO_2 levels have declined substantially over time, so has the importance of this impact (i.e. it is thought that current levels of SO_2 are unlikely to exceed an effect threshold). However, whether this assumption is correct or not is not known (Netcen, 2002).

On the economic valuation side, there is some literature on the economic value of cultural heritage sites. However, most of this literature is about significant changes in the quality of the sites or entry fees for visitors. Those studies that look at the damage from air pollution (covering only one or two specific sites) cannot be used in this context because of a lack of data on stock at risk (e.g. number of (similar) culturally important buildings, their surface areas, number and size of statues, repair and maintenance costs).

Climate change damage

The literature reports a wide range of economic values for damage costs of climate change. Damage costs refer to the monetisation of world-wide impacts of sea level rise and extreme weather events as well as impacts on human health, agriculture, water resources and ecosystems resulting from anthropogenic production of greenhouse gases. Damage cost models link emissions of greenhouse gases to changes in ambient concentrations to changes in global temperature to sea level rise and resultant damage.

Research into the economic impacts of climate change is hampered by uncertainties, not least regarding the understanding of climate change itself. While acceptance that global climate change is occurring is growing, current climate change scenarios and impact studies remain crude, and aggregation of costs requires many assumptions and value judgements. These factors all contribute to a low confidence in the numerical results of aggregate studies. Nonetheless, costs of climate change damages should continue to be an important factor in policy and programme appraisal.

Carbon dioxide

The Government's most recent formal recommendation on the unit value for costing the global warming damage from carbon dioxide emissions comes from an extensive review of the literature conducted by Clarkson and Deyes (2002), which concludes:

‘Therefore in terms of UK policy design, a point estimate for the social cost of carbon emissions of £70 per tonne of carbon could be used as an illustrative value, with associated sensitivity range with a lower bound of £35 [£9.5/tCO_2] and an upper bound of £140 [£38/tCO_2], for emissions in 2000. The point estimate should then be raised by £1 [£0.27/tCO_2] for each subsequent year.’

The report arrived at this estimate by adopting the values in Eyre et al (1998), which it judged to have used the most sophisticated methods to date. Several new studies on the social costs of carbon have been completed since that date, including a meta-analysis by Tol (2003), which suggests a lower value than £70 per tonne of carbon. There are also a number of impacts that are currently missing from the existing analyses which may increase valuations of social cost of carbon (SCC). Taking these points into consideration we recommend using the full range as suggested in the government’s recent formal recommendation of a range of £35 to £140 per tonne C or **£9.5 to £38 per tonne CO_2.**

The Clarkson and Deyes review was taken forward in October 2003 by a Defra chaired Inter-departmental Group on the Social Cost of Carbon (IGSCC), which is due to report findings in the summer of 2004. The group commissioned two research consortia to improve upon the current Government estimates, provide a better understanding of the uncertainty that surrounds them and to explore how to apply them to policy assessment. In the meantime, the group recommends that the above-mentioned range be employed in any cost-benefit analysis, applying caution where the balance may be tipped by the use of the upper or lower bound estimates.

Methane

The conventional approach to calculating the social cost of other greenhouse gases is to adopt global warming potentials (GWPs) and simply convert £X tCO_2 into £aX tCO_2eq for other GHGs where ‘a’ is the relevant GWP. However, using GWPs does not produce the correct ratios for the different GHGs in terms of the *marginal damage* done. This is largely due to the use of discounting in marginal damage estimates, whereas GWPs do not allow for discounting. In the current context it is marginal damage that matters. Despite this, the Kyoto Protocol requires that Parties use pre-defined GWPs to secure carbon-equivalents for the other GHGs.

Table 4.8 presents the results of various economic analyses of the time weighted impacts of the global warming effects of methane compared to carbon dioxide. These ratios are presented in the second column of the table. The range of possible values for the external costs of methane is then derived using these ratios from the literature and multiplying them by the Defra recommended estimates of the SCC of £9.5 - £38 per tonne of CO_2. In the last row of the table these ratios are compared with the UK Climate Change Programme’s recommended global warming potential for methane of 21 . The results indicate a range of values for the external cost of methane of £105 - £802/tonne CH_4.

Table 4.8 Estimates of the external cost of methane (£, 2003)

Study Ref	Ratio costs of CH_4 to CO_2	Based on Defra recommended SCC (£/tonne CO2) Low = 9.5	Medium = 19	High = 38
		Social Cost of Methane (£/tonne CH4)		
Tol (1999) [a]	14	133	266	532
Tol and Downing (2000)	19	181	361	722
Kandlikar (1995, 1996) [b]	12	114	228	456
Fankhauser (1995) [c]	20	190	380	760
Hammitt et al (1996) [d]	11	105	209	418
GWP	21	200	401	802

Notes:

[a] Emissions between 1995 and 2004; time horizon: 2100; discount rate: 3%; model: *FUND*1.6; scenario: IS92a; simple sum; no higher order effects.

[b] Time horizon: 100 years; discount rate: 2%; scenario: IS92a; quadratic damages.

[c] Emissions between 1991 and 2000; time horizon: 2100; GDP is calculated as a ratio of mean marginal damages.

[d] Emissions in 1995; time horizon: 2100; discount rate: 3%; scenario: IS92a; middle case.

Source: Tol and Downing (2002)

Another approach to arriving at a recommended value is to adopt only the values for the social costs of methane estimated in the Eyre et al (1998) study, which are based on marginal damage costs rather than global warming potentials. Adopting the methane values from this study provides consistency with the Government's approach to recommending a value for SCC which also uses this study to derive its values. Converting the marginal damage costs for CH_4 estimated in this study, and using the same approach as taken by Clarkson and Deyes[10], gives £315 per tonne of CH_4 (£, 2000) as a central estimate. In line with the approach to using SCC (i.e. halving the best estimate of £315/t CH_4 to provide a lower bound and doubling it to provide an upper bound) this presents a range of **£158 to £630 per tonne of CH_4**. This is the value we use as the recommended range in this study.

Note that if the Government's recommended estimate for SCC of £70/tC (£19/t CO_2) is converted to methane using the GWP of 21, this would result in a central estimate value of £401/t CH_4, which lies well within the range recommended here.

10 The Eyre et al (1998) figures were converted to 2000 prices using an inflation rate of 1.35, reversing out discounting to 1990 using a 3% discount rate, and converting from a dollar value using an exchange rate of $1=£0.56. The final figure is the average of the values suggested by the two models, Open Framework and the Fund.

Damage to agriculture

As with the estimation of the impact on building materials, there are three components to estimating the economic cost of the effects of air pollution on crop and forestry yields: establishing the dose-response function (i.e. kg of yield loss due to air pollution)[11], the estimation of the stock at risk, and unit damage costs.

Details of dose response functions and estimation of stock at risk are not covered in this section. With regard to the unit damage costs, this is determined by crop yield loss valued at the world market price of the crops concerned. The most up-to-date estimate of unit costs and total crop damage due to air pollution is likely to come from the ongoing work for Defra that will be updating the estimates of crop damage due to air pollution. These estimates will be included within unit damage cost estimates per tonne of pollutant, where ozone is regarded as the main pollutant of concern.

Other damage to ecosystems (flora/fauna)

Air pollution damages to ecosystems are dominated by the acidifying effects of pollutants like sulphur dioxide and nitrogen oxides on aquatic plants, fish, amphibians and freshwater invertebrates and on terrestrial ecosystems (forests, agriculture, moorlands, mosses and lichens, mammals and birds).

There are many economic valuation studies on the value of damages to ecosystems (see Mourato (1997) for an overview of the economic valuation literature of air pollution impacts on ecosystems). Apart from some impacts on agriculture already described above, the values in the literature are characterised by very large uncertainty ranges and weak dose response functions. The major shortcoming of the literature is a lack of scientific research to provide a relationship between the emission of pollutants (i.e. per kilogram) and their physical impacts on the environment.

Estimates of the acidifying effects of SO_2 on water ecosystems are provided in both Ecotec (1994) and AEA Technology (1998). Both reports base the valuation of damage to ecosystems on one study by Stale Navrud (1988), which estimates the WTP to reduce SO_2 emissions in order to protect salmon and trout ecosystems in Norway. Ecotec and AEA Technology both derive (different) simple relationships between WTP and reductions in SO_2 emissions. However, the study authors admit that these relationships are subject to major uncertainty and numerous assumptions, which make these estimates inappropriate for benefits transfer to the UK. For further details on these studies see Appendix 6.

More recent studies, notably ESRC (2003) on mountain lake ecosystems and acid rain deposition, will update the literature with

11 The consensus so far is that the following crops are sensitive to air pollution: wheat, barley, potato, sugar beet, rye and oats.

respect to lake ecosystems, but the issue of missing links between dose and response remains. Thus, there is little in the form of primary research that we can recommend for the purposes of this study at the time of writing.

4.2.2 Emissions to water

The scientific study distinguishes between emissions to sewer and emissions to surface water. The environmental impact of these releases, however, is only described in qualitative terms and not quantified.

The economic valuation literature for the water environment provides monetary estimates for some potential impacts but does not link these to pollutants; dose-response functions for the water environment being especially difficult to generalise. Thus, it is difficult to recommend valuation estimates without further analysis of the impacts from MSW waste management on the water environment. One exception is leachate, which merits further discussion (see below).

A component of the economic cost of water pollution from MSW can be found in the costs incurred by water companies to remove pollutants from water. The total amount that water companies in England and Wales spent on water treatment in 2002 was £439million (OFWAT, 2003). However a breakdown by pollutant type is not available. Research undertaken by Pretty et al (2000) estimate expenditure for nitrates and phosphates (two of the pollutants reported in the scientific study). For nitrate removal between 1992 and 1997 capital expenditure was estimated to be £18.8million/year and operating expenditures £1.7million/year. Capital expenditure by water companies on phosphate removal between 1992 and 1997: £68.8million/year and operating expenditures £4.3million. In order to begin to apply these estimates to the outputs of the scientific study, an understanding is required of quantity of nitrates and phosphates removed during this period of time. Whilst estimates could be derived such a calculation has not been undertaken in this study due to limited study resources.

Leachate

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. Contaminants can enter groundwater and/or surface water, where they can affect human health and the ecosystem. Valuation studies covering this impact have frequently used a clean-up costs approach (e.g. CSERGE et al, 1993). This approach is however not recommended for valuing external costs as it is not a true measure of the value of damage. At best it can provide a lower estimate of the expected range of values.

WTP studies have been undertaken on the value of clean water but this work is not relevant here for two reasons. Firstly, it is not possible to link the WTP results to any units of leachate from landfill, i.e. there is no direct association with landfilling, and secondly the context of the valuation study is different to that relating to leachate contamination and is therefore not directly transferable.

There are two studies that attempt to value impacts of leachate:

- USA study, Sharefkin et al (1984), which applies dose-response functions in order to find the damage to human health due to leachate in Price landfill in New Jersey, USA. The authors conclude that the damage would be in the range of \$22.8million to \$262million (\$,1984).

- Roberts et al (1991), which apply the contingent valuation method to estimate the external costs of a landfill in Knox County, Tennessee. The study found, inter alia, that households whose drinking water supplies were at risk of contamination were willing to pay \$141 more per year than those who used piped city water or bottled water.

Given that these studies are from the USA and are more than 10 years old, their results are not recommended for use in the current context. They also focus on human health as the endpoint of interest and there are many possible environmental receptors at risk from water pollution.

4.3 Disamenity

Some disamenity impacts of landfills are associated with the mere existence of landfill (stock or fixed externalities) and some are associated with the operation of the landfill as they vary with the amount and type of waste (variable or flow externalities). Whether related to stock or flow externalities, the term disamenity includes impacts such as noise, dust, litter, odour, presence of vermin, visual intrusion and the perception of risk to human health due to proximity to landfill.

Two valuation approaches have been used in the literature to estimate the disamenity impacts of landfills and incinerators. These are contingent valuation (a subset of stated preference techniques) and the hedonic price method (a subset of revealed preference techniques). Contingent valuation may or may not be explicit about the specific disamenity impacts being valued. In principle, using hedonic pricing methods, one can statistically identify the contribution of the bundle of disamenity impacts (e.g. noise, odour, litter and so on) to house prices/rental values, but typically sufficiently detailed data are not available to permit separate hedonic price function for each type of disamenity impact.

Thus, hedonic price studies usually use the distance between the property and the landfill as a proxy for the bundle of disamenity impacts. Which disamenity impacts are included in this bundle depends not so much on the actual existence of each but more on individual property owners' / buyers' perception of these impacts. In addition, in principle, there may be a potential for double counting if disamenity estimates are used in conjunction with the estimates of other impacts. An example of this is health impacts due to airborne pollutants from the operation of a landfill versus the element of 'risk to health' impact that could be perceived as part of the disamenity impact and captured by the hedonic pricing method. While there is no concrete evidence as to what is and is not perceived by property owners and buyers as part of the bundle of disamenity impacts, based on the general lack of public's understanding of environment-human health impact pathways, we suggest that the likelihood of such double counting is slight. The rest of this section looks at the available evidence on the economic costs of disamenity from landfills and incinerators.

4.3.1 Disamenity from landfills

Hedonic pricing

The most up-to-date and UK specific, and hence recommended, study on the disamenity costs of landfills was published by Defra in 2003. This study, conducted by Cambridge Econometrics, eftec and WRc, was a hedonic assessment of housing prices around landfill sites. The primary objective of the study was to estimate the effects of landfill disamenity experienced by households located at different distances from landfill sites.

The study finds that the total cost of disamenity due to landfills in Great Britain (at end of 1995) is £2,483million at 2003 prices representing a reduction of over £5,500 in the average value of a house lying within the zone of 0.25 miles from an operational landfill site and about £1,600 for those in the zone of 0.25-0.5 miles from an operational landfill site. Unlike the earlier (US) studies the study does not find any effect of landfills on property prices beyond the 0.5 miles mark. The same result can also be expressed as the disamenity cost per facility at £406,670 per operational landfill site (£670,000 in 2003 prices) (with a confidence interval of £334,350 to £478,990 in 2001 prices or £551, 000 to £789,000 in 2003 prices). Note that all the estimates above (total, per property, per facility and per tonne of waste) are expressed in present value terms due to the nature of the property prices.

The same disamenity cost can be expressed as £ per tonne of waste, by converting these stock values to annual equivalents and then dividing by the national annual throughput of waste. The study estimates this to be between £1.52 and £2.18 per tonne (£,2000) assuming average flow of 100million tonnes pa for 28 years and a discount rate of 6%. Adjusting these values to 2003 equivalents using housing price index changes, gives a range of **£2.50** to **£3.59** per tonne, with a central estimate of £3.05/tonne waste (£, 2003)**.**

There are a large number of other hedonic pricing studies in the literature, all of which take place in the US. Given the appropriateness of the Defra study to the purposes of the assessment here, these other studies are not reported.

Stated preference

A choice experiment study was undertaken by Willis and Garrod (1997) in order to estimate the WTP to reduce noise, odour and windblown dust and litter from a landfill. The study was rather small interviewing only 79 residents around the chosen landfill (Crawcrook Quarry and Landfill). The results can be summarised as follows (£,2003):

- Marginal WTP to reduce the number of days when respondent suffers from dust and windblown litter from the site: £0.12 to £0.19 per day;
- Marginal WTP to reduce the number of days when respondent can smell the site from their home: £0.10 – £0.15 per day; and
- Noise was not judged to be a significant problem.

In order to make the results of Willis and Garrod (1997) comparable to the Defra study, these marginal WTP estimates need to be aggregated across the UK population residing in the vicinity of all landfills and across time depending on the assumption of number of days (in a year) the impacts are felt by each household. Given that Defra report is the most relevant and recent study, it is not necessary to undertake this aggregation process which would have to be based on a number of assumptions and hence be inevitably weaker.

In addition, there are a number of stated preference studies about the disamenity from landfill sites in the US. But again, in the presence of a recommended UK study, these are not reviewed here.

4.3.2 Disamenity from incinerators

There is only one relevant study on the disamenity impacts of incinerators, Kiel and McClain (1995) an hedonic pricing study in Massachusetts, USA. The study examines the progression of house prices over the life of an incinerator, from rumour and construction and through to operation. It finds that after four years of operation the effect on house prices within 3.5 miles of the site has stabilised at £10,055 per mile (£,2003), which is essentially a distance decay function, i.e. starting from 3.5 miles from the site, at every mile approaching the site house prices drop by an average of £10,055.

This result has been converted to £/tonne of waste based on information on the case study site that could be gleaned from publicly available data (see Appendix 5 for details of this calculation). The outcome of the analysis is that £10,055 per mile is equivalent or **£21/tonne of waste** for the disamenity effect on local communities. This was arrived at by multiplying the per mile estimate by the stock of houses in each distance band, which gives approximately £7,800,000 per year for the incinerator. When this is compared to the average disamenity cost of landfill from the Defra study of between £551, 000 to £789,000 when converted to £,2003 prices using house price index changes, the results of Kiel and McClain seem disproportionately large. For this, and also given the fact that the study is from the US, this estimate is not recommended for use in the UK context.

Table 4.10 provides a summary of disamenity estimates for landfills and incinerators.

Table 4.10 Summary values of disamenity impacts from primary studies

Impact from Scientific Study	Primary Economic Study Reference	Units	Low	High	Best estimate
			£,2003		
Landfill (hedonic pricing method)	Defra (2003)	£ per facility	551,000	789,000	670,000
		£ per tonne of waste	2.50	3.59	***2.50 – 3.59***
Incinerator (hedonic pricing method)	Kiel and McClain (1995)	£ per facility	Not available	Not available	7.8million
		£ per tonne of waste	Not available	Not available	***21*** [a]

Note. [a] Not recommended for benefits transfer to a UK context.

4.4 Summary economic values by pollutant from primary studies

4.4.1 Emissions to air

This Section presents the results of work to estimate aggregate economic valuations per mass of pollutant. The data presented in this Section are taken from an independent piece of analysis undertaken by AEA Technology on behalf of Defra to support this economic study. The basis of AEA's analysis is the forthcoming study for Defra to assess the UK's Air Quality Strategy (referred to in this report as the Air Quality Strategy Evaluation report). [12] This analysis provides the most up to date and accurate analysis of the value of health and environmental impacts from waste management facilities when expressed on a mass of pollutant basis[13].

In order calculate external values per mass of pollutant from waste management facilities, this bespoke analysis has undertaken its own dispersion and dose response modelling. The functions used in this modelling and associated assumptions will be different from those applied in the scientific study to calculate health effects of waste

12 AEA Technology, Evaluation of the Air Quality Strategy – a study for Defra (Environment Policy Economics) forthcoming.

13 Although this is the best currently available information, the scientific and economic theory underlying this analysis is constantly evolving. Hence, these values may be altered as knowledge gaps are filled.

management facilities. This difference therefore introduces a potential source of error when using these valuation estimates in the context of the results from the scientific study. That said, such errors will be small relative to the best previously available figures, i.e. the BeTa work. Therefore, this bespoke analysis is more appropriate for use in the context of UK waste management facilities than the BeTa work because:

- The current modelling uses the most up-to-date estimates of mortality and morbidity – these are largely the same as those described earlier in this Section (more details are provided below).

- Specific modelling has been undertaken to reflect the exposure of populations around incinerators and landfills from emissions. As noted above, whilst this may not be precisely the same as that undertaken in the scientific study, it is substantially more appropriate than the BeTa approach which was focused on power sector and transport emissions did not include values for urban stack emissions or low level rural emissions.

Methodology

The following text describes the approach to modelling the impacts of PM10 emissions from incinerators. No description has been provided by the authors on how other pollutants have been modelled but it is assumed that a similar approach is used to that of PM10 along with appropriate dispersion and dose response functions. More details of the methodology are provided in the forthcoming Air Quality Strategy Evaluation report.

The analysis of impacts arising from PM10 emissions from incinerators is consistent with that used for other analyses presented in the Air Quality Strategy Evaluation report and with previously published work that have focussed on PM10 as a pollutant.[14] Contributions to ground level PM10 concentrations are influenced by emissions from point sources, area sources, secondary particles and coarse particles. Emissions from point sources specifically were modelled using two atmospheric dispersion modelling techniques. For large PM10 sources, (sources with >100 tonnes PM10 emission per annum) impacts of emissions were modelled using the ADMS 3.1 dispersion modelling package. For smaller point sources, (sources with <100 tonnes PM10 emission per annum), impacts were modelled using a dispersion matrix approach. In 2001, all incinerators in the UK fell into the latter category and therefore the impacts of emissions from incinerators will have been modelled with the dispersion matrix approach.

The technique assumes that each source emits into a nominal 1km by 1km by 50 metre volume and represents metrological conditions throughout the UK using 10-year statistical data for the Whatnall weather station 1985-94. A total of 789 point sources were modelled using this approach, 15 of these sources were MSW incinerators.

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14 Stedman et al, 2002.

The model outputs were combined with population data to provide population weighted PM10 concentrations. This was combined, in turn, with concentration-response functions to estimate the health impacts, and with monetary endpoints to calculate the economic values for total PM10 related air pollution for the UK.

The study also assessed the UK's National Atmospheric Emissions Inventory (NAEI) forecasts of incinerator emissions in 2005. Incinerator emissions are forecast to grow to 309 tonnes of PM10 by 2005. The NAEI spatially allocates these future emissions, again on an individual plant-by-plant basis, across the UK.

The study then looked at the potential impacts of increases in incinerator PM10 emissions. The analysis looked at the incremental effects of extra emissions from incinerators, both within one year, and over time. Six model runs were undertaken to look at a 10%, 50% and 100% increase in incineration emissions, in both years (2001 and 2005). This allowed the examination of the linearity of the marginal increase in emissions, and other modelling effects from changing background levels in the different years. In all cases, it was assumed that the increase in incineration emissions occurred as a proportionately uniform increase across all UK incinerator sites.[15]

The incremental health impacts estimated by model runs, over and above the baseline, were linked to the incremental emissions, to estimate a cost per tonne of PM10. The results, which are based on the cost per tonne for incremental emissions in 2001, were considered the most representative from a policy perspective and hence used in this study. The variation between a 10% and 100% increase in emissions in that year was found to be very low (i.e. increasing emissions by a small or large amount gave a very similar cost per tonne value).

The values quoted reflect a range that is based on different assumptions about the quantification of health impacts (for chronic mortality) and assumptions on the appropriate monetary endpoints for both acute and chronic mortality. The range is referred to as 'central low' and 'central high' estimates. It is stressed that this range is a restricted range estimated by altering only one or two parameters. It does not reflect the full range of values from all endpoints, nor the full statistical range associated with the overall methodology (and the uncertainty associated with emissions projections, modelling, health impact assessment and valuation).

The key assumptions used in the central low and central high values are summarised in Table 4.11 below.

15 In practice, the marginal increase in population weighted exposure, from any increase in incineration emissions at a given plant or from a new plant, will vary with the location of emissions, due to the strong influence of local population exposed to the pollution. It was concluded that the most representative approach for this study was to base incremental emissions on current incinerator sites as a whole.

Table 4.11 Assumptions used in the Defra Air Quality Strategy Evaluation Report

Category	Central low	Central high
Valuation of deaths brought forward	£3,100 (lower value in EAHEAP, adjusted upwards for inflation)	£110,000 (central value in ExternE, also value reported in EAHEAP for 1 year VsL)
Quantification of chronic mortality (years of life lost)	Use of PM2.5 (Pope functions, risk factor 1.01%)	Use of PM2.5 (Pope functions, risk factor 1.03%)
Valuation of chronic mortality (life year lost)	£31,500 (indicative value from forthcoming Defra valuation study)	£65,000 (central value in ExternE)
Discounting of chronic health effects	1.5% discount rate, based on a 1 year pollution pulse	1.5% discount rate, based on a 1 year pollution pulse
Threshold of effect	None, concentration-response functions implemented linearly, without threshold	None, concentration-response functions implemented linearly, without threshold
Value per tonne of carbon (£/tonne C)	£70	£70

Results

The results from this modelling are presented in Tables 4.12 and 4.13 for emissions of PM10, SO_2, NO_x and VOCs from landfill and incineration, respectively. The values shown are in £ 2003 prices per tonne of pollutant released. In the modelling it is assumed that landfills are predominantly in rural areas and incinerators in urban areas.

Note that CO_2 and methane are not included in these tables. The impact of these gases would need to be added to those shown here when determining the aggregate impact of emissions from landfill and incineration.

Table 4.12 Estimates of £/tonne of pollutant from UK landfill (£2003)

	Coverage	Central Low	Central High
PM10	***Health effects only***	***161***	***1,025***
SO_2		***643***	***2,941***
Of which health	Health effects only	418	2,716
Of which materials	Materials	225	225
NO_x	***Health effects from secondary pollutants, but excluding ozone***	***154***	***977***
VOC		***263***	***665***
Of which health	Health effects, including ozone	3	405
Of which crops	Crop damage, including ozone	260	260

Table 4.13 Estimates of £/tonne of pollutant from UK incineration (£2003)

	Coverage	Central Low	Central High
PM10	***Health effects only***	***6,119***	***39,245***
SO_2		***643***	***2,941***
Of which health	Health effects only	418	2,716
Of which materials	Materials	225	225
NO_x	***Health effects from secondary pollutants, but excluding ozone***	***154***	***977***
VOC		***263***	***665***
Of which health	Health effects, including ozone	3	405
Of which crops	Crop damage, including ozone	260	260

These results should be interpreted with the following qualifications in mind:

- The figures only include costs that occur in the UK - all trans-boundary pollution and impacts are excluded.
- Values for NO_x and SO_2 include secondary particulate (PM10) formation (nitrates and sulphates).
- Values for VOC include ozone formation and related effects, but values for NO_x do not.

- The numbers exclude several categories of impacts, notably:
  - Effects of NO_x on ozone formation (Note, ozone effects from NO_x could be positive as well as negative, due to issues with local NO + ozone reactions, and regional precursor levels)
  - Ecosystems effects (acidification, eutrophication, etc)
  - Effects on cultural or historic buildings from air pollution
  - Chronic mortality health effects from PM10 on children
  - Chronic morbidity health effects from PM10
  - Morbidity and mortality from chronic (long-term) exposure to ozone
  - Change in visibility (visual range)
  - Effects of ozone on materials, particularly rubber
  - Non-ozone effects on agriculture

Whilst the values provided by the Air Quality Strategy Evaluation report provide us with the best available estimates for the economic costs of air pollutants from waste management options, and the results are consistent with the VOSL recommended in this report, the numbers are subject to residual uncertainty particularly in respect to the quantification and valuation of chronic mortality.

The remaining air pollutants are: HCl/HF, 1,1-Dichloroethane, Chloroethane, Chlorothene, Chlorobenzene, Tertrachloroethene, Arsenic, Benzene, Cadmium, Nickel, Mercury, Dioxins and Furans, Polychlorinated Biphenyls. Valuation studies for these air pollutants are limited to their health impacts, i.e. cancer and deaths brought forward, as identified in the scientific study and covered in Section 4.1. Thus, to determine per kg estimates per pollutant, these health impacts need to be multiplied by the number of cancer cases and deaths brought forward per kg of pollutant.

4.5 Power generation

The terms of reference for this study requested an analysis of the avoided damage costs of power generation that result from energy recovery under certain waste management options. The most up-to-date reference on the subject is the Air Quality Strategy Evaluation report referred to above which applies the unit damage costs to weighted average emissions from the UK generating mix. The results are shown in Table 4.14. A significant feature of these figures is the high contribution of carbon emissions to the total. As noted in Section 4.2.1 the ranges in estimates of the external costs of CO_2 emissions are large. Hence there are substantial ranges in the figures for the central low and central high scenarios shown in Table 4.14 depending on the assumed social cost of carbon.

The external costs per kWh are proportional to the rate of release of emissions from the electricity supply industry. External costs will therefore decrease with expected reductions in UK electricity supply industry emissions. Some care must, therefore, be taken in using the 2001 numbers for future policy analysis.

The assumptions on unit value of health impacts are the same as those described for the Air Quality Strategy Evaluation report above.

Table 4.14 External costs of air pollution from power generation in the UK (pence/kWh)

2001			
	Range for Cost of Carbon		
Scenario	£35 per t/C	£70 per t/C	£140 per t/C
Central Low	0.59	1.03	1.91
Central High	1.17	1.61	2.49
Of Which Carbon	0.44	0.88	1.76
2005			
	Range for Cost of Carbon		
Scenario	£35 per t/C	£70 per t/C	£140 per t/C
Central Low	0.44	0.81	1.55
Central High	0.69	1.06	1.80
Of Which Carbon	0.37	0.74	1.48

5. Summary results

The following tables draw together the valuation estimates obtained from the primary studies. In these tables we have recommended values to allow quantitative integration with the scientific study. These recommended estimates are to be treated with caution, as there are still large uncertainties over the values. We strongly recommend, therefore, that when comparing the economic impacts from MSW management options that a full sensitivity analysis is conducted to explore implications of the uncertainties in these values.

5.1 Health impacts

Table 5.1 shows the summary values attributable to health impacts. These are all obtained from the primary studies identified in Section 4.

For deaths brought forward the full range of values taken from the key reference (EAHEAP, 1999) is £3,100 to £1.6m. However, as explained in Section 4.1.1 the most appropriate range for the value of deaths brought forward as a result of air pollution is £3,100 to £110,000.

For respiratory hospital admissions the same range in values as shown in Table 4.7 is recommended. Since the distribution of values within the range is not known, it is not possible to provide a single "central" estimate.

For cardiovascular hospital admissions, a single estimate is shown as an initial figure/range. This estimate is derived from point estimates of the willingness to pay to avoid the illness, the cost of foregone earnings and cost of hospitalisation. No data on the uncertainty around this estimate are given in the literature, but given the basis on which this estimate was made a range of +/- 50% would not seem inappropriate. This gives a recommended range of £1,337 to £4,011.

The individual values for leukaemia (£2,260,000) and lung cancer (£1,480,000) are based on Markandya (1997) plus our own calculations to include the cost of illness and forgone earnings. The recommended estimates for general cancers (£1,150,000 to £2,260,000) take into account a broader range of values associated with both types of cancer.

In the case of birth defects there are many types of defects but the scientific study does not make this differentiation. Therefore, as a general estimate for all birth defects, therefore, the full range of values is provided. To produce a narrower range or even a single central estimate we would need to have some understanding of the distribution of values within each category of defect and the likely weighting of types of defects around waste management facilities. In the absence of such information, rather than make unsubstantiated guesses on these points, the approach taken is to simply offer a broad range of recommended estimates and leave any decisions on weighting or averaging to the user of this information. Such decisions are likely to be sensitive to the context in which the calculations are made. The recommended range is therefore ***£90,500 - 423,000*** (£,2003)***.***

Table 5.1 Summary values for health impacts (£, 2003)

	Recommended Estimates	Initial Range / Figure
Deaths brought forward (£ per death brought forward)	***3,100 – 110,000***	***3,100 – 1,600,000***
Respiratory admissions (Casualty & Hospitalisation) (£ per admission)	***550 – 1,260***	***550 - 1,260***
Cardiovascular hospital admissions (£ per admission)	***1,337 – 4,011***	***2,674***
Cancers (£ per cancer case)		
Lung	1,480,000	53,000 – 1,480,000
Leukaemia	2,260,000	199,000 – 2,260,000
Haemangiosarcoma	cannot recommend	no value available
All cancers	***1,150,000 – 2,260,000***	***53,000 – 2,260,000***
Birth defects (£ per birth defect case)		
Neural tube defects	246,000 - 422,000	246,000 - 422,000
Cardiovascular defects	224,000 - 423,000	224,000 - 423,000
Hypospadias and epispadias	cannot recommend	no value available
Abdominal wall defects	148,000	148,000
Gastroschisis and exomphalos	90,500	90,500
Low birth weight	cannot recommend	no value available
Very low birth weight	cannot recommend	no value available
All birth defects	***90,500 - 423,000***	***90,500 - 423,000***

5.2 Air impacts

The results for air impacts are taken directly from Section 4.4 above. In this summary section we have also added in the costs of methane and CO_2 as key air pollutants.

Table 5.2 Summary estimates of £ per tonne of pollutant from UK landfill (£2003)

	Central Low	Central High
PM10	***161***	***1,025***
SO_2	***643***	***2,941***
Of which health	418	2,716
Of which materials	225	225
NOx	***154***	***977***
VOC	***263***	***665***
Of which health	3	405
Of which crops	260	260
CH_4	***158***	***630***
CO_2	***9.5***	***38***

Table 5.3 Summary estimates of £ per tonne of pollutant from UK incineration (£2003)

	Central Low	Central High
PM10	***6,119***	***39,245***
SO_2	***643***	***2,941***
Of which health	418	2,716
Of which materials	225	225
NO_x	***154***	***977***
VOC	***263***	***665***
Of which health	3	405
Of which crops	260	260
CH_4	***158***	***630***
CO_2	***9.5***	***38***

5.3 Water impacts

For the reason explained in Section 4.2.2 we have not quantified the value of releases to water from MSW management processes.

5.4 Disamenity impacts

Table 5.4 summarises the value of disamenity impacts from MSW landfill and incineration facilities in the UK. This is a repeat of Table 4.10. As noted in the footnote to the table, however, only the disamenity effects of landfill are suitable for application to UK MSW management processes. This clearly makes a comparison of landfill and incineration from a disamenity point of view incomplete.

Table 5.4 Summary values of disamenity impacts from primary studies

Impact from Scientific Study	Primary Economic Study Reference	Units	Low	High	Best estimate
			£,2003		
Landfill (hedonic pricing method)	Defra (2003)	£ per facility	551,000	789,000	670,000
		£ per tonne of waste	2.50	3.59	***2.50 – 3.59***
Incinerator (hedonic pricing method)	Kiel and McClain (1995)	£ per facility	Not available	Not available	7.8million
		£ per tonne of waste	Not available	Not available	***21*** [(a)]

Note. [(a)] Not recommended for benefits transfer to a UK context.

6. Literature gaps

In general, there are four reasons why there are gaps in the literature estimating the economic costs:

1. *Physical impact data may be missing*. This involves the gaps in the dose-response functions and should be covered by the scientific study. In general, dose-response functions for more studied impacts such as those of the acidifying pollutants are available. The impact pathways of other pollutants such as airborne heavy metals and contaminants in water are less well developed.

2. *Physical impact data may exist but may not be in a form that could be used for economic valuation.* Sometimes referred to as the 'correspondence' problem, this is especially an issue for complex environmental impacts that are generally expressed in units of, say, a pollutant, while what is needed (and missing) is the resulting impact that can be perceived by society and hence can be valued in economic terms.

3. *Economic research may be lacking.* For example, research on the effects of groundwater pollution on human health and the environment in general is not as well developed as, say, research on the effects of air pollution on human health. In the context of the current study, this may not be a significant gap. The impact of waste management options on groundwater is through leachate. We can argue that the technical design of landfills, where leachate is a potential problem, will improve so that risk of leachate will be minimised. However, a bigger gap in the context of this study is in terms of the economic costs of cardiovascular illnesses and reduced birth weight (around landfills).

4. *Economic research may exist but may not be clearly applicable to the context of this study.* For example, there are a number of WTP studies that cover types of cancers (e.g. skin and prostate) that are not associated with the environmental impacts of waste management options.

In addition to these general reasons for gaps in the literature, there is currently a temporary gap with regards to the context of this study. Examples include the mortality, agriculture and buildings and materials impacts of acidifying pollutants and the impacts of climate change. These gaps do not arise due to any of the above reasons but due to the nature of the research. As information about impact pathways, dose-response functions and people's preferences for the resulting impacts change, the methodologies are updated.

An interesting example of another gap in the literature is the effect of acidifying pollutants on ecosystems. Previous work on this was heavily dependent on the weak assumptions made about the impact and transferring WTP estimates from studies in different contexts and hence suffering from the above reasons for gaps in the literature. Appendix 6 provides an overview of work to date on the economic value of the acidifying effects of air pollution, and critiques the use of these values in the context of this study. The results of the ESRC (2003), a CVM study which estimates WTP to reverse the acidifying effects in remote mountain lakes, fill the gap in the literature to a certain extent for aquatic ecosystems. However, its results cannot be used with ease since the link between environmental impacts and the pollutants causing them is missing. This represents the opposite of the 'correspondence problem' referred to in point 2 above.

Finally, disamenity impacts of MRF are another gap in the literature. If they can be assumed to be similar enough to the disamenity impacts of landfill or incinerators, current economic cost estimates for these waste management options can also be used for MRF. However, if the physical impacts are deemed to be significantly different, new economic research will have to be conducted.

There are a number of studies that are forthcoming that will add to the existing literature. Relevant to this study are:

- *Valuation of Health Benefits Associated With Reductions in Air Pollution (Chilton et al, 2004).* This study has been in progress for nearly two years and is investigating how much people in the UK are willing to pay for reductions in the health risks associated with air pollution. It comprises an extensive UK contingent valuation survey (665 interviews) to determine values for loss of life from acute and chronic mortality, respiratory hospital admissions and other breathing difficulties.

- *International comparison of mortality valuations (Markandya et al, 2004).* The study aims to compare WTP estimates of the value of acute and chronic mortality valuations in the US, France, Italy and the UK using the same methodology. The UK partner is the University of Bath

7. References

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Appendices

1. Profile of secondary studies

Publication details

Authors	COWI (2000)
Title	A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste
Commissioning Organisation	European Commission, DG Environment.

Environmental focus

This report was commissioned to review studies which give a shadow price of disposing of waste in an incinerator and a landfill. The report concentrates on seven studies with a European focus, identifies the assumptions underlying the shadow price and evaluates each study's methodology.

The study concludes that whilst the methodology supporting the valuation of air-pollution is becoming established, the pollution pathways and dose-response functions of pollutants released to soil and water are subject to greater uncertainty.

Shadow prices

In the majority of case studies the environmental valuation was given per metric of externality. For emissions the original value is typically given per kilogram of emissions. For amenity values the value is expressed as a change in amenity per day, or a percentage increase in house prices per km.

Sources of shadow price information

Source of Data	Ref#	Geographical Coverage	Method and Impacts	Original Sources of Valuation data
Rabl et al(1998)	4	European Focus	Morbidity and mortality impacts only. Includes coefficients for population density and stack height.	Based on ExternE for primary (NO_x, SO_2, particles, VOC and CO_2) pollutants and other studies for secondary pollutants (heavy metals).
ETSU (1996)	3	Based on three incinerators from around Europe	This includes morbidity, mortality, damage to buildings and crops.	Based on ExternE for primary pollutants and the US for damage functions.
Tellus Institute (1992)	Not Cited	US	The specific environmental impacts can not be disaggregated.	Uses a control cost approach to environmental legislation. Therefore, the environmental value is determined by existing legislation.

Source of Data	Ref#	Geographical Coverage	Method and Impacts	Original Sources of Valuation data
ECON (1995)	2	Norway	For the primary pollutants (NO_x, SO2, particles, VOC and CO_2) the environmental value includes morbidity, mortality damage to buildings. For secondary pollutants (heavy metals) it only includes the health impacts	Based on various Norwegian and International studies for the primary pollutants, uses tax levels for CO_2 and uses an index linked to lead for secondary pollutants.
Coopers & Lybrand and CSERGE (1996)	Referred to separately			
Miranda and Hale (1997)	6	Sweden, Germany, UK and the US	Includes health impacts and other environmental impacts but the results for each plant are not disaggregated.	
Garrod and Willis (1998)	8	UK	Disamenity costs associated with a landfill site in the NE of England.	A study of the value of a reduction in visible debris and odour externalities.
Brisson and Pearce (1995)	7	US	Literature survey of the hedonic pricing approach.	

Publication details

Authors	Coopers & Lybrand, CSERGE and eftec (1996)
Title	Cost-Benefit Analysis of the Different Municipal Solid Waste Management Systems: Objectives and Instruments for the Year 2000
Commissioning Organisation	European Commission

Environmental focus

The objective of the study was to project the total net economic cost from 1993 until 2001 of MSW treatment in each EU member state. 31 different waste management methods were compared according to each country's economic characteristics and waste arisings.

The total economic value was estimated according to the life-cycle emissions associated with a particular disposal route. These were then used to calculate an expected shadow price. As a result of the life-cycle emissions used in the study, the shadow price calculated of each disposal route depends primarily on assumptions about transport and displaced electricity generation rather than the emissions associated with the waste arising.

Shadow prices

For a selected group of externalities, SO_2, NO_x and PM, this study provides a shadow price per unit of externality in each country. The country's shadow price is determined by the level of deposition into the sea, and an estimate of each country's willingness to pay. The report also estimates the externalities associated with a specific waste disposal technique, based on the life-cycle analysis. The shadow prices given for these disposal techniques are therefore given per tonne of waste generated.

Sources of shadow price information

Source of Data	Ref #	Geographical Coverage	Method and Impacts	Original Sources of Valuation data
ExternE	30	Europe		
Fankhauser	24	Global	Literature survey and estimation of the environmental cost of climate change.	

Publication details

Authors	CSERGE, eftec, Jaakko Poyry (1995)
Title	The British Newsprint Manufacturers Association, Recycling or Incineration – an evaluation of the alternatives
Commissioning Organisation	The Paper Federation (GB)

Environmental focus

The study compared the impact of disposing of newsprint with incineration and with recycling. Four analyses were undertaken using existing and best available technologies. These analyses were:

- Life-Cycle Analysis - to evaluate the environmental efforts of recycling and incineration and provided the physical emissions estimates.
- National Economic Analysis - to evaluate the impact on employment, value added and the balance of payments.
- Social Cost Benefit Analysis - to introduce an economic value to environmental emissions and social impacts.
- Industry Competitiveness Analysis - a specific analysis of the newsprint and recycled fibre market.

The emissions factors are provided in the third study as kgs of pollutant per tonne of newsprint according to the source of material used in production (virgin or recycled material) and the disposal method (incineration, landfill or recycle). Where emissions are displaced, with energy generation, the displaced generator is assumed to be an old coal fired generator.

Shadow prices

The shadow prices used in this study were determined according to the type of pollutant being analysed. Pollutants to air are reported according to the type of pollutant and whether or not they include the economic value of damage caused outside of the UK. Pollutants to water are determined according to their clean up costs. This measure only refers to the chemical oxygen demand.

Sources of shadow price information

Source of Data	Reference #	Geographical Coverage	Method and Impacts	Original Sources of Valuation data
CSERGE / eftec (1993)	Referred to Separately			

Publication details

Authors	ERM and eftec (1999)
Title	Technical Review of Studies on the Externalities of Waste Management Options
Commissioning Organisation	Department of Environment Transport and Regions, UK

Environmental focus

This study examines eight reports which have examined the environmental impacts of waste management options. As a number of the reports reviewed in this study are also secondary sources we have reviewed them separately. A number of other reports included in this study are only relevant to a very specific field, or express the environmental values in a way which prevents them from being integrated into this report. The only report that is expanded upon is therefore the ETSU (1996) report to the European Commission.

The study concludes that many of the studies cited lack the coverage of emissions to provide a complete assessment of the lifecycle costs of different waste disposal techniques and that not all the noted burdens are given a shadow price, most notably disamenity.

Shadow prices

The shadow prices given in the study are normally provided per unit of externality, although some reports gave medical impacts. For the ETSU (1996) report, the units were € / tonne of emissions. The values were provided according to the assumed stack height of the incinerator, ranging from 50 to 100m, we have entered these values as the high and low points in a range.

Sources of shadow price information

Source of Data	Reference #	Geographical Coverage	Method and Impacts	Original Sources of Valuation data
CSERGE, Warren Spring Laboratory and eftec (1993)	Referred to Separately			
Coopers & Lybrand, CSERGE and eftec (1996)	Referred to Separately			
eftec and CSERGE (1998)	Referred to Separately			
ENTEC (1999)	Regulatory Impact Assessment of the Proposed Waste Incineration Directive	Not referred to directly as the report gives the total impact on human health rather than the economic cost or the type of health impact.		

Publication details

Authors	CSERGE and eftec (1998)
Title	Life Cycle Research Programme for Waste Management: Damage Cost Estimation for Impact Assessment
Commissioning Organisation	Environment Agency

Environmental focus

This report was part of a five phase research and development programme supported by the Environment Agency to develop a waste management decision tool. The purpose of this report was to develop a database of shadow prices relating to various externalities. The database was required to show the shadow price according to five receptor categories; human health, buildings, crops, climate change and ecosystems.

The report covered eight categories of externalities:

- Greenhouse gases;
- Ozone depleting substances;
- Air pollutants (SO_2, NO_x, PM10);
- Organic compounds (VOCs);
- Heavy metals (lead);
- Eutrophication;
- Transport (accidents, noise and congestion); and
- Disamenity.

However, they highlight the regional nature of disamenity and problems with developing a generic figure.

Shadow prices

Where possible the shadow prices are expressed in cost per kg of emission according to their specific impact. Rather than estimating the vehicle kilometres involved in a waste disposal technique the shadow price of transport is expressed per vehicle kilometre, and the shadow price of eutrophication is expressed as a price per annum for cleaner water.

Sources of shadow price information

Source of Data	Ref #	Geographical Coverage	Method and Impacts	Original Sources of Valuation data
Fankhauser	24	Global	Literature survey and estimation of the environmental cost of climate change.	
Eyre et al	26	Global	Study on the costs of global warming assuming a marginal damage estimate.	
Berry et al	27	Various Geographical Areas in the UK	Range of estimates for direct health damages resulting from major pollutant emissions.	Orignally based on VLYL but adapted for VOSL.
AEA Technology	28	UK	Estimates the repair and maintenance cost of different materials and environmental impacts of air pollution.	
Zylicz et al	29	Baltic	Estimates the benefits associated with reducing nitrogen and phosphorous.	Based on results by Sandstrom (1994), Zylicz (1995) and Soerqvist (1995).
DETR	31	UK	Traffic Accident Data and economic costs.	
Maddison et al	32		Surveys the economic cost of air-pollution related health impacts.	
Newbery	33	UK	Estimates of UK congestion costs and traffic.	Traffic model.

Publication details

Authors	CSERGE, Warren Spring Laboratory and eftec (1993)
Title	Externalities from Landfill and Incineration
Commissioning Organisation	Department of the Environment

Environmental focus

This report estimates the externalities associated with waste disposal to landfill and incineration. It then establishes a shadow price per tonne of waste disposed according to each disposal method. Five main categories of externalities are covered by the report:

- Greenhouse Gases, specifically Carbon and Methane
- Major air pollutants, these include SO_2, NO_x and Particulates
- Disamenity, as represented by hedonic prices and contingent valuation (CVM) studies
- Leachate – which is based on monitoring costs, and
- Associated Activities, such as displaced electricity generation and accidents.

Whilst the study's primary focus is the UK, and most of the data used is based on UK studies, some information is transferred from US studies particularly with regards disamenity.

Environmental values

The shadow prices attributed to greenhouse gases, and major air pollutants are expressed per tonne of emissions. These shadow prices were derived from either published studies or the author's own estimates based upon published studies.

The shadow prices attributed to leachate, and associated activities, are given per tonne of waste. Disamenity shadow prices are normally given according to the distance from the landfill, although this may change to a different metric in a CVM study.

Sources of Shadow Price Information

Source of Data	Refer-ence #	Geo-graphical Coverage	Method and Impacts	Original Sources of Valuation data
Frankhauser	24	Global	Literature survey and estimation of the environmental cost of climate change.	
Nordhaus	22	US	Uses the DICE model to estimate the cost of damage on the Economy.	
Peck & Teisberg	23	Global	Built the CETA model to derive the optimum level of control.	
Nelson, Genereux, Genereux	9	USA	Use the hedonic pricing approach to determine the increase in property prices for each mile from a landfill.	
Havlicek, Richardson and Davies	10	USA	Uses the hedonic pricing approach to determine the increase in property prices for each mile from a landfill.	
Havlicek	11	USA	Uses the hedonic pricing approach to determine the increase in property prices for each mile from a landfill.	
Alder et al	12	USA	Studies the change in house prices in a three mile radius following a water contamination incident.	
Gamble et al.	13	USA	Uses the hedonic pricing approach to determine the fall in property prices for each mile nearer to a landfill site.	
Mendelsohn et al	14	USA	Uses the hedonic pricing approach to determine the fall in prices within a 2 mile radius of a landfill site.	
Kohlhase	15	USA	Uses the hedonic pricing approach to determine the fall in property prices for each mile nearer to a landfill site.	
Michaels & Smith	16	USA	Establishes the average gain in house price with removal of landfill, using a Hedonic Pricing Approach.	
Baker	17	USA	Uses the hedonic pricing approach to determine the fall in property prices for each mile nearer to a landfill site	
Hirshfeld et al	18	Hypothetical	Uses the hedonic pricing approach to determine the fall in property prices for each mile nearer to a landfill site.	
Roberts et al	19	USA	Use a contingent valuation method approach to determine the value to residents of moving the landfill elsewhere.	
Smith et al	20	USA	Uses a contingent valuation method approach to determine the average consumer surplus per mile from landfill.	

Publication details

Authors	ExternE (1998)
Title	Externalities of Energy
Commissioning Organisation	DG Environment

Environmental focus

The ExternE project was established to assess the impacts of airborne pollutants on five receptors:

- Health (Morbidity and mortality)
- Crops and Agriculture
- Building Materials
- Forests, and
- Ecosystems

To achieve this ExternE has developed a European model of environmental impacts according to geographical area, dose-response functions and monetary costs attributable to each impact.

Shadow prices

The method for deriving a monetary cost attributable to each environmental externality varies according to the receptor impact. These have been derived through a variety of methods including

- Market Value – If an externality has an impact on the quantity of a good sold to market, for example crops or health care, then the market value can be considered.

- Associated market value – If an externality has an impact on the quality of a good sold to market, such as house prices, then a hedonic method can be used.

- Non-Market Value – If an externality has an impact on the quality or quantity of a good which can not be expressed directly, such as bathing water quality, then a contingent valuation or travel cost method can be used.

The ExternE project relies on other economic studies to report the economic cost of each externality. ExternE combines the economic cost with the geographical dose-response function to establish the total expected cost. This allows an estimated cost per unit of emissions to be developed which can be expressed either as a European or National average. It is noted however, that there will be considerable difference at the national scale according to population characteristics.

Sources of shadow price information

Not listed

Publication details

Authors	IER
Title	External Costs of Energy Conversion – Improvement of the ExternE Methodology and Assessment of Energy-Related Transport Externalities
Commissioning Organisation	European Commission, Joule III Programme

Environmental focus

The research was undertaken to review and update the underlying methodology of ExternE. Specifically these revisions included:

- Updating global warming impacts with the most recent climate change models
- Reviewing the global, regional and local impacts of Ozone
- Reviewing monetary valuation issues
- Updating exposure response models

With regards the monetary valuation specific updates have concerned:

- Value of a Life Year Lost (VLYL) – the update recommended changing the approach to mortality from a Value of Statistical Life (VOSL) to a VLYL, arguing that reduced the difference between the WTP and the QALY approaches.
- Morbidity estimates and endpoints - 3 studies have been released since the last ExternE report was reviewed.
- Non Marginal changes of costs – updates have introduced a methodology for extended partial equilibrium analysis or full general equilibrium analysis.
- Non-monetary environmental impacts – updates have provided guidance on using avoidance costs (where WTP monetary values are unavailable) and options for multi-attribute decision-making.

Shadow prices

The shadow prices presented represent the damage cost per tonne of pollutant emitted according to geographical area. The report presents national shadow prices for rural and local emissions from road and rail transport.

Sources of shadow price information

Source of Data	Reference #	Geographical Coverage	Method and Impacts	Original Sources of Valuation data
Otterstrom et al (1998)	39	Helsinki	Willingness to pay for better air quality	
Rozan (1999)	38	Strasborough	Willingness for improved health damage	
CSERGE et al (1999)	37	Netherlands, Norway, Portugal, Spain & UK	Benefits transfer and valuation of environmental damage	
Downing et al (1995)	44	European	Fossil fuel cycle assessment for ExternE	
Tol (1999b)	46	Global	Damage costs of climate change	

Publication details

Authors	NETCEN (2002)
Title	BeTa – Benefits Table Database, Estimates of the Marginal External Costs of Air Pollution in Europe
Commissioning Organisation	European Commission, DG Environment

Environmental focus

The BeTa database was compiled as a part of the ongoing improvements of the ExternE database. Using the same methodology as the ExternE database, the BeTa database addresses three specific sources of emissions:

- Emissions from all sources in rural locations
- Emissions at ground level in urban areas – e.g. traffic
- Emissions from shipping

The impact of these sources is assessed for PM2.5, SO_2, NO_x and VOCs over a 1000km radius.

Shadow prices

The report presents national shadow prices for rural and local emissions for transport and shipping emissions.

Sources of shadow price information

Source of Data	Reference #	Geographical Coverage	Method and Impact	Original Sources of Valuation data
ETSU (1997)	54	Europe	Cost benefit analysis of incineration of non-hazardous waste.	
Holland & Kin (1998)	55	Europe	Economic evaluation of air quality targets for tropospheric ozone.	
Holland et al (1999a)	56	Europe	Economic evaluation of emission ceilings for atmospheric pollutants.	
Holland et al (1999b)	57	Europe	Economic evaluation of limits for CO and Benzene.	
Faircloth et al (1999)	58	European Accession Countries	Benefits of compliance with the environmental acquis.	
Holland et al (1999c)	59	Netherlands	Cost benefit analysis for the protocol to abate acidification, eutrophication and ground level ozone.	
Brown et al (2000)	60	Europe	Economic evaluation of PVC waste management.	
Holland et al (2001)	61	Europe	Economic evaluation of PAHs (Polycyclic Aromatic Hydrocarbons).	

2. Summary of secondary studies

		Cost of Emissions to Air (£ 2003)							Cost of Emissions to Water (£ 2003)							Cost of Emissions to Land (£ 2003)							
Externality	Units	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Original Reference
1,2,3-Trichloropropane	£/tonne of Emissions	6,877,728							6,877,728							6,877,728							COWI (2000) Economic Valuation of Landfill Externalities
1,3 Butadiene	£/tonne of Emissions	70,479																					IER,(1999), External Costs of Energy Conversion
2-Butanone	£/tonne of Emissions	823,243							823,243							823,243							COWI (2000) Economic Valuation of Landfill Externalities
4-methyl 2-Pantanone	£/tonne of Emissions	823,243							823,243							823,243							COWI (2000) Economic Valuation of Landfill Externalities
Acetone	£/tonne of Emissions	406,411							406,411							406,411							COWI (2000) Economic Valuation of Landfill Externalities
Anitmony (Sb)	£/tonne of Emissions	102,749,088							102,749,088							102,749,088							COWI (2000) Economic Valuation of Landfill Externalities
Arsenic (As)	£/tonne of Emissions	862,008,573							260,520							10,421							COWI (2000) Economic Valuation of Landfill Externalities
Arsenic (As)	£/tonne of Emissions	1,213																					COWI (2000) Economic Valuation of Landfill Externalities
Arsenic (As)	£/tonne of Emissions		1,570,011																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Arsenic (As)	£/tonne of Emissions		11,478																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Arsenic (As)	£/tonne of Emissions		231,258																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Arsenic (As)	£/tonne of Emissions		171,004																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Barium (Ba)	£/tonne of Emissions	312,624							26,052							31,262							COWI (2000) Economic Valuation of Landfill Externalities
Benzene	£/tonne of Emissions	1,862																					IER, External Costs of Energy Conversion - Improvement of the Externe Methodology and Assessment of Energy-Related Transport Externalities, (1999) Bickel, Schmid
Beryllium (Be)	£/tonne of Emissions	38,035,920							38,035,920							38,035,920							COWI (2000) Economic Valuation of Landfill Externalities
bisphthalate	£/tonne of Emissions	239,678							20,842														COWI (2000) Economic Valuation of Landfill Externalities
Cadmium (Cd)	£/tonne of Emissions	106,396,368							4,328,139							1,281,758							COWI (2000) Economic Valuation of Landfill Externalities
Cadmium (Cd)	£/tonne of Emissions	61,045																					COWI (2000) Economic Valuation of Landfill Externalities
Cadmium (Cd)	£/tonne of Emissions	14,802																					COWI (2000) Economic Valuation of Landfill Externalities
Cadmium (Cd)	£/tonne of Emissions		125,431																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Cadmium (Cd)	£/tonne of Emissions		106,294																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Cadmium (Cd)	£/tonne of Emissions		697,594																				ERM, EFTEC (1999) Review of Externalities of Waste Management

		Cost of Emissions to Air (£ 2003)							Cost of Emissions to Water (£ 2003)							Cost of Emissions to Land (£ 2003)							
Externality	Units	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Original Reference
Carbon Dioxide (CO2)	£/tonne of of Newsprint						5																CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Carbon Dioxide (CO2)	£/tonne of Emissions						7																CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Carbon Dioxide (CO2)	£/tonne of Emissions						1																CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Carbon Dioxide (CO2)	£/tonne of Emissions						3																CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Carbon Dioxide (CO2)	£/tonne of Emissions						7																CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Carbon Dioxide (CO2)	£/tonne of Emissions						6																CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Carbon Dioxide (CO2)	£/tonne of Emissions						6																Pearce et al (1998), Lifecycle Waste Management
Carbon Dioxide (CO2)	£/tonne of Emissions						14																Pearce et al (1998), Lifecycle Waste Management
Carbon Dioxide (CO2)	£/tonne of Emissions						38																COWI (2000) Economic Valuation of Landfill Externalities
Carbon Monoxide (CO)	£/tonne of of Newsprint	7																					CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Carbon Monoxide (CO)	£/tonne of Emissions	2																					IER, (1999) External Costs of Energy Conversion
Carbon Monoxide (CO)	£/tonne of Emissions	2																					COWI (2000) Economic Valuation of Landfill Externalities
CFC 11	£/tonne of Emissions		7,546																				Pearce et al (1998), Lifecycle Waste Management
CFC 12	£/tonne of Emissions		6,185																				Pearce et al (1998), Lifecycle Waste Management
CFC 13	£/tonne of Emissions		6,792																				Pearce et al (1998), Lifecycle Waste Management
Chemical Oxygen Demand (COD)	£/tonne of of Newsprint												-0.038										CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Chemical Oxygen Demand (COD)	£/tonne of of Newsprint												0.001										CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Chromium (Cr)	£/tonne of Emissions	170,275,871							270,941							14,797,536							COWI (2000) Economic Valuation of Landfill Externalities
Chromium (Cr)	£/tonne of Emissions	99,488																					COWI (2000) Economic Valuation of Landfill Externalities
Chromium (Cr)	£/tonne of Emissions		1,239,884																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Chromium (Cr)	£/tonne of Emissions		2,321,070																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Copper (Cu)	£/tonne of Emissions	41,683							1,042							4,168							COWI (2000) Economic Valuation of Landfill Externalities
Diethypalate	£/tonne of Emissions	239,678							20,842														COWI (2000) Economic Valuation of Landfill Externalities
Dioxins (TEQ)	£/tonne of Emissions	605,240,061,626							53,187,762,991														COWI (2000) Economic Valuation of Landfill Externalities
Dioxins (TEQ)	£/tonne of Emissions	13,184,189,255																					COWI (2000) Economic Valuation of Landfill Externalities
Dioxins (TEQ)	£/tonne of Emissions		1,661,100,000																				COWI (2000) Economic Valuation of Landfill Externalities

		Cost of Emissions to Air (£ 2003)							Cost of Emissions to Water (£ 2003)							Cost of Emissions to Land (£ 2003)							
Externality	Units	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Original Reference
Eutrophication	£/Per annum of to improve the baltic Sea												70										Pearce et al (1998), Lifecycle Waste Management
Eutrophication	£/Per annum of to improve the baltic Sea												38										Pearce et al (1998), Lifecycle Waste Management
Eutrophication	£/Per annum of to improve the baltic Sea												119										Pearce et al (1998), Lifecycle Waste Management
Halon 1211	£/tonne of Emissions	38,483																					Pearce et al (1998), Lifecycle Waste Management
Halon 1311	£/tonne of Emissions	90,546																					Pearce et al (1998), Lifecycle Waste Management
Hydrogen Fluoride (HF)	£/tonne of Emissions	1,875,744																					COWI (2000) Economic Valuation of Landfill Externalities
Hydrogren Chloride (HCl)	£/tonne of Emissions	5,210																					COWI (2000) Economic Valuation of Landfill Externalities
Leachate	£/tonne of Landfilled												1										COWI (2000) Economic Valuation of Landfill Externalities
Leachate	£/tonne of Landfilled												0										COWI (2000) Economic Valuation of Landfill Externalities
Leachate	£/tonne of waste landfilled												1										CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Leachate	£/tonne of Emissions												1188116										COWI (2000) Economic Valuation of Landfill Externalities
Lead (pb)	£/tonne of Emissions	29,386,656														4,168							COWI (2000) Economic Valuation of Landfill Externalities
Lead (Pb)	£/tonne of Emissions	1,221,989																					Pearce et al (1998), Lifecycle Waste Management
Mercury (Hg)	£/tonne of Emissions	21,987,888							7,043,833														COWI (2000) Economic Valuation of Landfill Externalities
Methane (CH4)	£/tonne of of Newsprint						85																CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Methane (CH4)	£/tonne of Emissions						1,892																COWI (2000) Economic Valuation of Landfill Externalities
Methane (CH4)	£/tonne of Emissions						6																Coopers & Lybrand & CSERGE (1996), CBA of different MSW Management Systems
Methane (CH4)	£/tonne of Emissions						112																CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Methane (CH4)	£/tonne of Emissions						108																Pearce et al (1998), Lifecycle Waste Management
Methane (CH4)	£/tonne of Emissions						125																Pearce et al (1998), Lifecycle Waste Management
Methyl Chloride	£/tonne of Emissions	4,064,112							4,064,112							4,064,112							COWI (2000) Economic Valuation of Landfill Externalities
N20	£/tonne of Emissions			2,189																			Pearce et al (1998), Lifecycle Waste Management
N20	£/tonne of Emissions			5,395																			Pearce et al (1998), Lifecycle Waste Management

		Cost of Emissions to Air (£ 2003)							Cost of Emissions to Water (£ 2003)							Cost of Emissions to Land (£ 2003)							
Externality	Units	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Original Reference
Nickel (Ni)	£/tonne of Emissions	86,180,016							10,421							3,126							COWI (2000) Economic Valuation of Landfill Externalities
Nickel (Ni)	£/tonne of Emissions	13,704																					COWI (2000) Economic Valuation of Landfill Externalities
Nickel (Ni)	£/tonne of Emissions	2,046																					COWI (2000) Economic Valuation of Landfill Externalities
Nickel (Ni)	£/tonne of Emissions		24,781																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Nickel (Ni)	£/tonne of Emissions		228,288																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Nickel (Ni)	£/tonne of Emissions		22,829																				ERM, EFTEC (1999) Review of Externalities of Waste Management
Nitrates	£/tonne of Emissions							9,744															Pearce et al (1998), Lifecycle Waste Management
Nitrogen Oxides (NOx)	£/tonne of of Newsprint	900																					CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Nitrogen Oxides (NOx)	£/tonne of Emissions			1,534																			IER, (1999) External Costs of Energy Conversion
Nitrogen Oxides (NOx)	£/tonne of Emissions			2,037																			BeTa (vs. E1.02a) Benefits Table Database
Nitrogen Oxides (NOx)	£/tonne of Emissions			6,207																			ExternE
Nitrogen Oxides (NOx)	£/tonne of Emissions			5,106																			COWI (2000) Economic Valuation of Landfill Externalities
Nitrogen Oxides (NOx)	£/tonne of emissions			4,831																			CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Nitrogen Oxides (NOx)	£/tonne of Emissions	14,600																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Nitrogen Oxides (NOx)	£/tonne of Emissions	2,341			269																		Coopers & Lybrand & CSERGE (1996), CBA of different MSW Management Systems
Nitrogen Oxides (NOx)	£/tonne of Emissions	1,228	698	-135																			Pearce et al (1998), Lifecycle Waste Management
Nitrogen Oxides (NOx)	£/tonne of Emissions		29,091		140																		ERM, EFTEC (1999) Review of Externalities of Waste Management
Ozone (03)	£/tonne of Emissions	-624	-53	-208																			Pearce et al (1998), Lifecycle Waste Management
Ozone (03)	£/tonne of Emissions	767,791	48,347	471,956																			Pearce et al (1998), Lifecycle Waste Management
Ozone (03)	£/tonne of Emissions		2,147																				ERM, EFTEC (1999) Review of Externalities of Waste Management

		Cost of Emissions to Air (£ 2003)							Cost of Emissions to Water (£ 2003)							Cost of Emissions to Land (£ 2003)							
Externality	Units	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Mortality	Morbidity	Agriculture	Buildings	Ecosystem Impacts	Climate Change	Disammenity	Original Reference
Particulates (PM2.5)	£/tonne of Emissions	307,314																					IER, (1999) External Costs of Energy Conversion
Particulates (PM2.5)	£/tonne of Emissions	7,599																					BeTa (vs. E1.02a) Benefits Table Database
Particulates (PM2.5)	£/tonne of Emissions	1,427	654																				Pearce et al (1998), Lifecycle Waste Management
Particulates (PM10)	£/tonne of of Newsprint	10,944																					CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Particulates (PM10)	£/tonne of Emissions	12,827																					ExternE
Particulates (PM10)	£/tonne of Emissions	17,403																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Particulates (PM10)	£/tonne of emissions	1,832																					CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Particulates (PM10)	£/tonne of Emissions	24,917																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Particulates (PM10)	£/tonne of Emissions	11,000																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Particulates (PM10)	£/tonne of Emissions	6,171			212																		Coopers & Lybrand & CSERGE (1996), CBA of different MSW Management Systems
Particulates (PM10)	£/tonne of Emissions	10,098			212																		Coopers & Lybrand & CSERGE (1996), CBA of different MSW Management Systems
Particulates (PM10)	£/tonne of Emissions	2,622	1,339																				Pearce et al (1998), Lifecycle Waste Management
Particulates (PM10)	£/tonne of Emissions		226,469																				ERM, EFTEC (1999) Review of Externalities of Waste Management
p-Cresol	£/tonne of Emissions	125,050							10,421							83,366							COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Phenol	£/tonne of Emissions	104,206							10,421							10,421							COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Selenium (Se)	£/tonne of Emissions	13,651,248							13,651,248							13,651,248							COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Sulphur Dioxide (S02)	£/tonne of of Newsprint	2,043																					CSERGE, EFTEC, Jaakko Poyry, (1995) Recycling of Incineration in Newsprint
Sulphur Dioxide (S02)	£/tonne of Emissions	10,085																					IER, (1999) External Costs of Energy Conversion
Sulphur Dioxide (S02)	£/tonne of Emissions	3,525																					BeTa (vs. E1.02a) Benefits Table Database
Sulphur Dioxide (S02)	£/tonne of Emissions	6,492																					ExternE
Sulphur Dioxide (S02)	£/tonne of Emissions	1,772																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Sulphur Dioxide (S02)	£/tonne of emissions	9,984																					CSERGE, Warren Spring and EFTEC (1993), Externalities from Landfill and Incineration,
Sulphur Dioxide (S02)	£/tonne of Emissions	3,173																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Sulphur Dioxide (S02)	£/tonne of Emissions	9,868																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Sulphur Dioxide (S02)	£/tonne of Emissions		15,322		962																		ERM, EFTEC (1999) Review of Externalities of Waste Management
Sulphur Dioxide (S02)	£/tonne of Emissions	3,256		18	404	3																	Coopers & Lybrand & CSERGE (1996), CBA of different MSW Management Systems
Sulphur Dioxide (S02)	£/tonne of Emissions	5,776	658	10	370	13																	Pearce et al (1998), Lifecycle Waste Management

		Cost of Emissions to Air (£ 2003)							Cost of Emissions to Water (£ 2003)							Cost of Emissions to Land (£ 2003)							
Externality	**Units**	**Mortality**	**Morbidity**	**Agriculture**	**Buildings**	**Ecosystem Impacts**	**Climate Change**	**Disammenity**	**Mortality**	**Morbidity**	**Agriculture**	**Buildings**	**Ecosystem Impacts**	**Climate Change**	**Disammenity**	**Mortality**	**Morbidity**	**Agriculture**	**Buildings**	**Ecosystem Impacts**	**Climate Change**	**Disammenity**	**Original Reference**
Tin	£/tonne of Emissions	3,126																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Toluene	£/tonne of Emissions	10,421							1,042							1,042							COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Trans-1, 2-dichloroethylene	£/tonne of Emissions	2,084,160							2,084,160							2,084,160							COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Vanadium	£/tonne of Emissions	21,987,888																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Vinyl Chloride (VC)	£/tonne of Emissions	218,833							145,891							58,669,104							COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Volatile Organic Compound (VOC)	£/tonne of Emissions	797																					IER, (1999) External Costs of Energy Conversion
Volatile Organic Compound (VOC)	£/tonne of Emissions	1,488																					BeTa (vs. E1.02a) Benefits Table Database
Volatile Organic Compound (VOC)	£/tonne of Emissions	1,146																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Volatile Organic Compound (VOC)	£/tonne of Emissions	566																					COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Volatile Organic Compound (VOC)	£/tonne of Emissions			2,101																			COWI (2000), Economic Valuation of Environmental Externalities from Landfill
Zinc (Zn)	£/tonne of Emissions	6,252							1,042							1,042							COWI (2000), Economic Valuation of Environmental Externalities from Landfill

3. Profile of primary studies

Air pollution - morbidity		
Study reference		
Authors	Day, B., Dubourg, R., Machado, F., Mourato, S., Navrud, S., Ready, R.C., Spanniks, F. and Vazquez, M/X.	
Title	Non-contextual values for the avoidance of episodes of ill-health: Tests for the Stability of Benefits Across National Boundaries.	
Source of study	CSERGE, IOS-NLH, IVM, CAS and DAE-UoV. *Benefits Transfer and the Economic Valuation of Environmental Damage in the European Union: With Special Reference to Health*. Report to DG Environment	
Date of reference	1999	
Study area and human population characteristics		
Country	The Netherlands, Norway, Portugal, Spain and the UK	
Location	Amsterdam, Oslo, Lisbon, Vigo (Spain)	
Study population characteristics	Four of the surveys were conducted in cities, whilst a representative sample for all of the UK was used.	
Environmental focus of the study		
Specific environmental goods/services valued	Willingness to pay to avoid an episode of ill health caused by air pollution	
Extent of environmental change	Eyes:	1 day with mildly red, watering and itchy eyes. A runny nose with sneezing spells. Patient is not restricted in their normal activities.
	Cough:	1 day with persistent phlegmy cough, some tightness in the chest and some breathing difficulties. Patient cannot engage in strenuous activity, but can work and do ordinary daily activities.
	Stomach:	1 day of persistent nausea and headache, with occasional vomiting. Some stomach pain and cramp. Diarrhoea at least twice during the day. Patient is unable to go to work or leave the home, but domestic chores are possible.
	Bed:	3 days with flu-like symptoms including persistent phlegmy cough with occasional coughing fits, fever, headache and tiredness. Symptoms are serious enough that patient must stay home in bed for 3 days.
	Casualty:	A visit to hospital casualty for oxygen and medicines to assist breathing problems caused by respiratory distress. Symptoms include a persistent phlegmy cough with occasional coughing fits, gasping breathing even when at rest, fever, headache and tiredness. Patient spends 4 hours in casualty followed by 5 days at home in bed.
	Hospital:	Hospital admission for treatment of respiratory distress. Symptoms include persistent phlegmy cough with occasional coughing fits, gasping breath, fever, headache and tiredness. Patient stays in hospital receiving treatment for three days followed by 5 days home in bed.

Study methods	
Type of study	Primary study
Survey/study information	The survey consisted of 5 sections. The first section included questions about respondents' own health, i.e. whether they had asthma, bronchitis or respiratory allergies. How many days during the previous month they had experienced upper and lower respiratory symptoms and whether they had visited an emergency room or been admitted to hospital for respiratory problems in the last year. The second section asked respondents to rank the ill-health episodes in terms of how bad they were. The third section, respondents were asked to value the different episodes. The fourth section included attitude and behaviour questions and the fifth section collected standard socio-economic information.
Economic measure	Willingness-to-pay (WTP) to avoid an ill-health episode
Valuation technique	Stated Preference Technique: Contingent Valuation
Estimated values	
Results	The results showed that Norway and Spain tend to have significantly higher WTP to avoid ill-health episodes and the UK has significantly lower WTP. Overall WTP is higher for episodes that last longer. The three episodes that only last 1 day (cough, eyes and stomach), have the lowest mean WTP values in every country. Table 1 presents the results.

Table 1	**Mean WTP for health end-points: £,1997 per ill health episode (s.e.)**					
	Pooled	**Nether-lands**	**Norway**	**Portugal**	**Spain**	**UK**
Hospital	306 (13.41)	283	301	300	426	164
Casualty	158 (7.71)	128	239	185	146	131
Bed	97 (5.20)	71	119	88	113	83
Cough	27 (2.13)	28	36	28	39	20
Eyes	35 (2.94)	40	31	70	53	14
Stomach	35 (35.49)	n/a	n/a	61	n/a	26

Currency and unit of estimate	£(1997) per ill health episode
Information for aggregation	Local currencies were converted to UK £ 1997 by using the OECD estimates of purchasing power parity adjusted exchange rates.

Underlying assumptions	
Assumptions	Respondents were simply asked for their WTP to avoid the described episode of ill-health, without being told why they would suffer this episode or the means by which their payment would allow them to avoid it. This absence of context was designed to minimise intercultural differences in WTP and provide baseline figures that could be more readily transferred across national boundaries and across different policy contexts.
Appropriateness for benefits transfer	
Benefits transfer comment	The average error involved in transferring an estimate of WTP to a target country is 36%. This error was seen to increase to 44% when adjustment was made for socio-economic characteristics of the target population.

Abstract

This paper presents the main research findings from a contingent valuation study of morbidity resulting from exposure to environmental pollution in five European countries. The paper reports estimates of meant WTP to avoid each of six different ill-health episodes in each country.

A non-contextual survey instrument was employed since it was believed that this would minimise intercultural differences in WTP and provide baseline figures that could be more readily transferred across national boundaries and across different policy scenarios.

The results of tests on the feasibility of transferring these values across national boundaries are reported. A test of the reliability of a simple benefits transfer using unadjusted values, suggest an average error or 0.362. In other words, the error associated with simply transferring the values from this research to a country other than the five included in the study, would generate, on average, an over- or underestimate of 36.2%. When the transfer figures are adjusted for income the over- or underestimate is 44.2%.

Air pollution - morbidity	
Study reference	
Authors	Rozan, Anne
Title	Évaluation contingente des bénéfices de santé d'une amélioration de la qualité de l'air. L'exemple de la région Strasbourgeoise.
Source of study	Dissertation / thesis
Date of reference	1998
Study area and human population characteristics	
Country	France, Germany
Location	Strasbourg (France) and Kehl (Germany)
Study population characteristics	The population over 18 years of the 400,000 inhabitants in the community of Strasbourg.
Environmental focus of the study	
Specific environmental goods/services valued	Human health
Extent of environmental change	An improvement in air quality of 30%-50%
Study methods	
Type of study	Primary
Survey/study information	The sample is representative of the Strasbourg community by quotas of sex, age, occupation, residency and size of household. The one-to-one interview lasted between 10 and 20 minutes. The questionnaire included 3 parts. In the first part the respondent describes the symptoms of his suffering and the health state of his family. In the second part a program that leads to improvements in the air quality is described, and the respondent is asked whether he is willing to pay for the program and thus improve his health, or have no cost and no improvement. If the respondent is willing to pay, he is presented a contingent scenario in one of two versions (first 50%, then 30% improvement of air quality, or the opposite). The respondent also lists his reasons to participate in the program and to be willing to pay for it. The third part of the questionnaire includes social and socio-economic variables. A bracketing technique is used, with a low (50F) and a high (200F) bid.
Economic measure	Willingness-to-pay (WTP)
Valuation technique	Stated preference technique: Contingent valuation
Estimated values	
Mean (standard error)	The mean cost of pain was estimated at 300 FF. The average stated willingness to pay is 50% higher when respondents are informed in the questionnaire that air pollution is the origin of the bad health state. Every version of the questionnaire implied starting point bias. 55.3 % of the sample reported willingness to pay. Depending on whether or not the disease induced a sick leave, the weight of the cost of pain represents between 15% and 150% of the total cost.

Table 1	**Mean WTP for health improvement: FF**		
	No	**WTP**	**Test of 0-hypothesis**
Low version	16	205	Rejected (2%)
High version	16	698	Anchored
Low version	8	279	Rejected (3%)
High version	8	538	Anchored
Low starting point	16	263	Rejected (2%)
High starting point	16	825	Anchored
Note	Low version: the technique used in the contingent valuation that starts at 50 FF. High version: the technique used in the contingent valuation that starts at 200 FF. Low starting point: the technique that permits anchor testing at starting point 50 FF. High starting point: the technique that permits anchor testing at starting point 200 FF. 0-hypothesis: equality of means.		
Currency and unit of estimate	FF		
Information for aggregation			
Underlying assumptions			
Assumptions			
Appropriateness for benefits transfer			
Benefits transfer comment	Benefits transfer is not recommended on the basis that an internal test comparing the results between the German and the French samples showed very different results. Further, it is not clear what health effects of air pollution are reduced.		
Abstract			
The thesis focuses on the evaluation of health benefits of improved air quality. The principal result is that individual costs (i.e. the disutiliy associated with the morbidity effects) are at least as important as the collective costs (medical care, drugs, hospital, sick leaves). The contingent valuation method is used for assessing the private costs. The study is carried out in Strasbourg in France and in the neighbour town Kehl in Germany, so the results between these two populations are compared and the benefit transfer method is tested. The two surveys are carried out in the same period, with the same questionnaire and in areas with the same pollution problem. Nevertheless a benefit transfer is not acceptable. The Germans seem to be more sensitive to environmental problems than the French. The paper shows that expressed willingness to pay for reducing symptoms caused by air pollution, crucially depends on the respondents being aware of the origin of the symptoms. Providing the information secures better control over individual responses.			

Disamenity - landfill	
Study reference	
Authors	Cambridge Econometrics, eftec, WRc
Title	A Study to Estimate the Disamenity Costs of Landfill in Great Britain
Source of study	Department for Environment, Food and Rural Affairs (Defra)
Date of reference	2003
Study area and human population characteristics	
Country	UK
Location	All areas of Great Britain, except Northern Ireland
Environmental focus of the study	
Specific environmental goods/services valued	Fixed disamenity from UK landfill sites, including: noise, dust, litter, odour, presence of vermin, visual intrusion and enhanced perceptions of risk.
Extent of environmental change	Household proximity to landfill site: 0-0.25, 0.25-0.5, 0.5-1, 1-2 miles.
Study methods	
Type of study	Primary study
Survey/study information	Data on landfills taken from the England and Wales Environment Agency 1993/4 based database and the Scottish Environment Protection Agency, 1995 database: 11,300 landfill sites considered. Data for the property market were taken from: Cambridge Econometrics Area Housing Price Differential, Office for National Statistics and the Post Office postcode directory: In total, 592,000 housing transactions between 1991-2000 inclusive were considered.
Economic measure	Quantity Compensating Surplus (QCS)
Valuation technique	Hedonic property price technique.
Estimated values	
Mean values	Part of the explanation for house prices being lower in the vicinity of landfill sites comes from the clustering of low-income households in lower quality housing. Controlling for both the physical and socio-economic factors there remained a statistically significant stock disamenity effect for houses located 0-0.5 miles from a landfill site. Specifically there was a £5,500 reduction in house value for houses 0.25km from landfill site and a £1,600 reduction in house value for houses 0.25-0.5km from landfill site. Mean total present value of fixed disamenity: £2,483m, 95% confidence interval: £2,041m-£2,925m. (2001 price) Present value of fixed disamenity from landfill sites: £406,670 per landfill, 95% confidence interval: £334,350 - £478,990 per landfill. (2001 price)

	£1.86 / tonne of landfill, 95% confidence interval: £1.52-£2.18 / tonne of landfill. (2001 price) Table 1 presents the per cent change in house prices according to distance from landfills across the different regions of Britain. Table 1 shows there are marked regional variations. Scotland shows the largest stock disamenity effects. This may reflect data differences with the distribution of houses dominated by Glasgow and Edinburgh and with houses undergoing transactions on average located much closer to landfill sites than elsewhere across rural Scotland and Britain as a whole. Table 2 presents the variation in disamenity by type of landfill site. Table 2 shows that different types of waste have different fixed disamenity effects. Special/hazardous waste sites show similar negative disamenity effects as the 'all sites' estimates. By contrast, 100% biodegradable or 100% inert sites show no statistically significant negative disamenity effects. Table 3 presents the present value of landfill disamenity in Britain.

Table 1	**% change in house prices according to distance from landfill**				
Location	**Miles from landfill**				
	0-0.25	**0.25-0.5**	**0.5-1.0**	**1.0-2.0**	**2+**
South West	1.11	-0.05	0.43	-0.04	0.00
West Midlands	-1.45	-2.73	-0.98	0.24	0.00
South East	-1.25	-0.55	-0.58	-0.15	0.00
East	-4.88	5.55	1.5	0.06	0.00
East Midlands	-10.01	-8.75	20.93	9.89	0.00
Yorks + Humber	0.45	-1.22	1.38	2.16	0.00
North West	-1.52	-0.88	2.92	0.50	0.00
North East	0.7	1.07	0.96	1.82	0.00
West	-1.15	-1.13	0.46	-0.94	0.00
Scotland	-41.27	-7.73	-3.01	-2.67	0.00
South England	-1.39	0.63	0.03	-0.01	0.00
North England	-0.34	-0.53	2.01	1.14	2.23
Wales + Midlands	-5.39	-5.65	5.89	3.66	0.00
Great Britain	-7.06	-2.00	1.04	0.7	0.00

Table 2	**Variation in disamenity costs by type of site: GB**				
Distance from house (miles)	**Sample size**	**mean**	**Standard deviation**	**Standard error**	
All sites					
0-0.25	13638	-0.048	0.300	0.0039	
0.25-0.5	39669	0.003	0.452	0.0027	
0-0.5	53307	-0.010	0.419	0.0023	
0.5-1	81475	0.033	0.584	0.0024	
1-2	95398	0.030	0.517	0.0022	
100% special waste					
0-0.25	30	-0.018	0.142	0.0548	
0.25-0.5	98	0.016	0.160	0.0457	

0-0.5	128	0.008	0.156	0.0370	
0.5-1	383	0.008	0.188	0.0299	
1-2	546	0.014	0.235	0.0221	
100% biodegradable					
0-0.25	388	0.015	0.221	0.0154	
0.25-0.5	1049	0.044	0.683	0.0144	
0-0.5	1437	0.036	0.595	0.0113	
0.5-1	1641	0.057	0.679	0.0146	
1-2	52	0.144	0.229	0.0717	
100% inert					
0-0.25	5794	0.020	0.651	0.0068	
0.25-0.5	14380	0.088	0.984	0.0062	
0-0.5	20174	0.068	0.901	0.0049	
0.5-1	22004	0.033	0.545	0.0044	
1-2	3763	0.043	0.620	0.0087	

Table 3	**Present value of landfill disamenity in Great Britain**					
Distance from nearest landfill	**% reduction in price**	**Av. price £, 1995**	**Reduction in price: £ 1995**	**% of sample**	**No.of houses:**	**PV(£m) 2001 prices**
0-0.25	-7.06	69,807	-4,927	1.0	246	1,356.9
0.25-0.5	-2.00	70,546	-1,410	2.9	713	1,126.4
0.5-2	0	75,222	0	26.9	6616	0
2+	0	77,064	0	69.2	17021	0
Currency and unit of estimate	£ reduction in house value; £ / landfill; and £/ tonne of average waste to landfill.					
Underlying assumptions						
Assumptions	Present value calculations assume average flow of 100million tonnes of co-disposable waste material for 28 years at 6% discount rate.					
Appropriateness for benefits transfer						
Benefits transfer comment	On the basis that the QCS estimates are based on underlying consumer preferences they are not specific to a particular market. Under the assumption that preferences are stable across geographical regions, the demand functions can be transferred across markets. Data required to transfer these results includes information on: income, socio-economic characteristics and the proposed change in attribute levels to be experienced in the transfer process.					

Abstract
This study identifies and estimates the disamenity costs of landfill in Britain. These are the costs associated with local nuisance caused by landfill activity, including, noise, odour, litter, vermin, visual intrusion and any perceived discomfort. The hedonic property price technique is used to estimate the effect of proximity of landfill sites on house prices for particular types of landfill sites across Britain. Controlling for physical and socio-economic factors evidence of a statistically significant negative stock disamenity effect for houses located closer than 0.5 miles from a landfill site was found. The analysis showed considerable regional differences and evidence that different types of waste have different fixed disamenity effects.

Disamenity - landfill	
Study reference	
Authors	Guy Garrod and Ken Willis
Title	Estimating Lost Amenity Due to Landfill Waste Disposal
Source of study	Resources, Conservation and Recycling published by Elsevier
Date of reference	1997
Study area and human population characteristics	
Country	UK
Location	Crawcrook Quarry and landfill site at Crawcrook near Gateshead in Tyne and Wear in the North East of England
Study population characteristics	Local population: 9184 (1991 Census), Residents familiar with the effects of the landfill site Five landfill sites in rural areas of Gateshead borough
Environmental focus of the study	
Specific environmental goods/services valued	Disamenity effects associated with an established landfill site, including: noise, litter and odour disamenity.
Extent of environmental change	Reduction in number of days per year: i) noise disturbance from the site and its traffic; ii) when you can smell the site from your home; and iii) windblown dust and litter from the site landing in your property.
Study methods	
Type of study	Primary study
Survey/study information	Gravel has been extracted from the site for the last 50 years, and worked out areas of the quarry have been used for landfill since the late 1980s. The site disposes of non-toxic household, commercial and industrial wastes (paper, plastic, rubble, ash, garden waste etc). It receives up to a maximum of 1,200 tonnes per day (roughly 120 lorry loads). Approximately 400 households lie on its eastern and southern sides and the main lorry route. Over 100 houses were targeted in a random sample. A questionnaire was delivered to the occupants with an explanation of the study. Respondents were left at least 2 days to complete the survey which included general questions relating to issues of household waste, opinions about waste disposal issues and opinions about the Crawcrook landfill. Once completed, the interviewer invited respondents to take part in the choice experiment exercise. Overall, the sample size was 73, with each respondent making 4 choices, thus giving 292 observations.
Economic measure	Willingness-to-pay (WTP)
Valuation technique	Stated preference technique: Choice experiment
Estimated values	
Results	Overall, it was found that people living close to the landfill site were used to the disamenity levels associated with the site and had learnt to accept them. This meant there was little incentive to pay for measures to reduce existing levels of disamenity generated by the site. Noise was the least significant and there was no evidence respondents would be willing to pay to reduce

	current noise levels. The problem of dust and litter was found slightly more important than odour but in both cases WTP amounts to reduce current levels were low. The WTP amounts are presented below: • Marginal WTP to reduce number days when respondent suffers from dust and windblown litter from the site: £0.11-£0.18 per day. • Marginal WTP to reduce number days when respondent can smell the site from her home: £0.09 – £0.14 per day.
Currency and unit of estimate	£ per day, £ per year.
Information for aggregation	WTP for 50 fewer days with smells from the site and 50 fewer days with windblown litter from the site: up to £13 per year.
Underlying assumptions	
Assumptions	Respondents are assumed to have chosen the alternative that offered her the highest utility.
Appropriateness for benefits transfer	
Benefits transfer comment	The results are suitable for benefits transfer to a well established landfill site. However, the results are not considered suitable for estimating the disamenity impacts generated by the establishment of a new landfill site. In such instances the initial welfare losses could be great, but would diminish as residents become inured to the effects of the site. At present there is no evidence available to estimate the rate at which disbenefits decrease.
Abstract	
The increasing quantities of waste produced by society have implications not only for waste management but also for the social and environmental impacts that these activities generate. This paper examines the impacts that a well-established United Kingdom landfill site has on the people who live around it and uses a stated preference choice experiment to estimate the magnitude of these impacts in monetary terms. It is found that many residents experience minimal impacts having learnt to live with the landfill site. Willingness to pay for reducing impacts is relatively low and any reduction in landfill operations is likely to have little effect on the amenity of local people.	

Disamenity - incineration	
Study reference	
Authors	Katherine A. Kiel and Katherine T. McClain
Title	House Prices during Siting Decision Stages: The Case of an Incinerator from Rumour through Operation.
Source of study	Journal of Environmental Economics and Management 28, 241-255.
Date of reference	1995.
Study area and human population characteristics	
Country	United States
Location	North Andover, Massachusetts (20 miles north of Boston)
Study population characteristics	The data set consisted of 2593 single family home sales, from January 1974 to May 1992. North Andover is located near several major highways and has a total area of 27.85 square miles. Only houses within North Andover are included because tax treatment, city services and host benefits are unique to the town. The population was 99% white until 1990 when it dropped to 97%. The population has grown from 10980 in 1960 to 16247 in 1970, 20129 in 1980 to 22792 in 1990. Education levels have remained fairly stable over the period. The town has a single school district with four elementary schools, one middle school and one high school.
Environmental focus of the study	
Specific environmental goods/services valued	The effects of a new incinerator plant on residential property values over time.
Extent of environmental change	The effect of a new incinerator on residential property prices over 5 siting stages: Pre-rumour 1974-78, Rumour: 1979-80, Construction: 1981-84, Online: 1985-88 and Ongoing Operation: 1989-92.
Study methods	
Type of study	Primary
Survey/study information	The data contained structural information about each house, including the sales price, the number of bedrooms, baths, floor space, age of house. Other information included: whether the house had a lakefront, neighbourhood, distance from the incinerator, entrance to a major highway and the central business district. Information on occupants of the house could not be obtained. Prices are adjusted to control for the regional trend in sales prices over the period. The dependent variable used is the natural log of the sales price in current dollars divided by an index based on the median sales price of the existing single-family homes in the area. The study involved a separate regression for each of the five stages.
Economic measure	
Valuation technique	Income Capitalisation Modelling

Estimated values	
Results	The coefficient that measured the impact of the incinerator on house prices is not significant in the pre-rumour stage, this suggests that the eventual site of the plant was not inherently undesirable before it was selected for the incinerator. The coefficient was also not significant in the rumour stage. This means that house prices did not respond to the negative publicity the proposed facility was receiving and the public opposition voiced. By the time construction began, houses further from the site received a premium. The marginal impact of distance on the value of a house in North Andover during the construction phase is estimated to result in a change in the nominal sales price of \$0.43 per foot or \$2283 per mile. This premium paid for distance persisted through the early operating years (online) and is estimated at \$8100 per mile. After 4 years of operation, the premium decreases to \$6607 per mile. The persistence of a premium indicates that either the incinerator is viewed as a permanent disamenity or full adjustment takes longer than assumed in this study. Overall, the benefits of increasing the distance from the incinerator are maximised at 20000ft (3.5 miles)
Currency and unit of estimate	\$ per foot from incinerator, \$ per mile from incinerator
Underlying assumptions	
Assumptions	The minimum and maximum distance between the incinerator and a house were assumed 3500ft and 40000ft, respectively.
Appropriateness for benefits transfer	
Benefits transfer comment	The implications of this study are relevant beyond the effects of siting an incinerator plant. Any locally unwanted land use, such as a landfill site or recycling plant has the potential to affect local house prices although the profile of the price changes through time will differ with the type of facility. Although it may not be advisable to transfer the unit value results to a UK context due to significant differences in housing and population characteristics, the method for estimating price profiles could be followed.
Abstract	
The impact of an undesirable land use on house prices is not constant over time. Previous proximity studies which employed only discrete changes in information, such as an EPA Superfund site announcement, ignored potentially important phases of the adjustment process. This study explicitly measures how the effects of an undesirable land use evolve over the siting process and life of the disamenity. Some price response to rumours of a facility is indicated and the evidence that prices respond at groundbreaking, before operation, is strong. The distance premium persists at least 7 years after the facility begins operations.	

Other - supply of aggregates	
Study reference	
Authors	London Economics
Title	The Environmental Costs and Benefits of the Supply of Aggregates
Source of study	DETR Report
Date of reference	1999
Study area and human population characteristics	
Country	UK
Location	Various locations (21 sites + national survey)
Study population characteristics	Aggregate results are representative of England, Wales and Scotland.
Environmental focus of the study	
Specific environmental goods/services valued	The environmental impacts of the supply of aggregates, including disamenity at sites (eg lorry traffic, noise and vibration, dirt, dust, impacts on environment + recreation opportunities, visual intrusion, use of old quarries for waste disposal, danger for children, impacts on property prices, etc) and at national parks (eg visual intrusion, noise, water pollution, impacts on plants and wildlife, lorry traffic).
Extent of environmental change	The survey values the elimination of all of the above impacts through closure of quarry sites and restoration of the landscape.
Study methods	
Type of study	Primary
Survey/study information	A survey of 21 sites (sample size = 9361) plus a national survey (sample size = 1019) were conducted using face-to-face interviews at respondents' residences. A payment card elicitation format was used, and extensive debriefing. The survey was pre-tested with focus groups and pilots. Respondents were provided with maps indicating the location of quarry sites, and in the case of national parks were provided with example photos of the landscape with the quarries.
Economic measure	Willingness to Pay
Valuation technique	Contingent Valuation
Estimated values	
Results	National Average WTP per tonne of aggregate (£,1998) • Local survey – residents responses for hard rock sites not located in National Parks: £0.34 • Local survey – residents responses for sand and gravel sites: £1.96 • National survey – non-residents responses for quarries in National Parks: £10.52
Currency and unit of estimate	£ per tonne of aggregate
Information for aggregation	Aggregation already performed.

Underlying assumptions	
Assumptions	
Appropriateness for benefits transfer	
Benefits transfer comment	These figures can be successfully applied to tonnes of aggregate replaced by use of bottom ash in construction.
Abstract	
This high-quality CVM study used a large-scale survey (with over 10,000 responses) that assesses how much people would be willing to pay to stop the external environmental effects of quarrying for construction aggregates, both in their locality and in landscapes of national importance. The research was mainly undertaken to inform decisions about the regulation and possible taxation of the aggregates industry.	

Other - Congestion		
Study reference		
Authors	David Newbery	
Title	Economic Principles Relevant to Road Pricing	
Source of study	Oxford Review of Economic Policy, 6(2).	
Date of reference	1992	
Study area and human population characteristics		
Country	United Kingdom	
Location	National	
Study population characteristics	Drivers and passengers	
Environmental focus of the study		
Specific environmental goods/services valued	Value of time Road transport	
Extent of environmental change	Marginal cost of congestion on different types of road due to HGV and private car traffic	
Study methods		
Type of study	Primary	
Survey/study information	No survey was involved. Information about the methodology from a number of references listed in the article.	
Economic measure	Marginal congestion cost	
Valuation technique	Marginal congestion cost using value of time. There are three steps to this valuation exercise. The first is to estimate a relationship between speed of a car and the flow of traffic on the road. From this, in second stage, we can calculate the reduction in speed (increase in journey time) due to an additional vehicle on the road (increase in flow). Finally, the increased journey time is valued at the current value of time.	
Estimated values		
Results	The results show the marginal time costs of congestion in Great Britain in 1990 prices. The unit is pence per vehicle km. The results are presented in terms of passenger car units (PCU). The cost per HGV used in the main report is assumed to be 1.75 times that of the PCU.	
Table 1	Marginal Time Costs of Congestion	
Type of road	Pence per HGV km	Pence per PCU km
Motorway	0.46	0.26
Urban central peak	63.65	36.37
Urban central off-peak	51.15	29.23
Non-central peak	27.76	15.86
Non-central off peak	15.30	8.74
Small town peak	12.06	6.89
Small town off-peak	7.35	4.20
Other urban	0.14	0.08
Rural dual	0.12	0.07

carriageway		
Other trunk and principal	0.33	0.19
Other rural	0.09	0.05
Weighted average	6	3.4
Currency and unit of estimate	In 1990 prices	
Underlying assumptions		
Assumptions	Calculations are based on the assumptions about traffic flows, speed and value of time estimates current in 1990.	
Appropriateness for benefits transfer		
Benefits transfer comment	Estimates could be out of date but no other more recent estimates exist.	
Abstract		
Vehicles impose four main costs on the rest of society – accidents, environmental pollution, road damage and congestion. This article looks at road damage and congestion costs. There are three steps to estimating the marginal cost of congestion. The first is to estimate a relationship between speed of a car and the flow of traffic on the road. From this, in second stage, we can calculate the reduction in speed (increase in journey time) due to an additional vehicle on the road (increase in flow). Finally, the increased journey time is valued at the current value of time.		

4. Economic conversion calculations

In transferring WTP estimates from another country to the UK, two adjustments are usually needed. The first adjustment (spatial) is to convert from the other currency to the UK pound sterling. The second adjustment (temporal) is to convert from an earlier year's currency to the year of concern (in the case of this report: £,2003).

There are two approaches to spatial adjustment. The first is using the purchasing power parity (PPP) between the two countries. PPP is a theory which states that exchange rates between two currencies are in equilibrium when their purchasing power is the same in each of the two countries. This means that the exchange rate between two countries should equal the ratio of the two countries' price level of a fixed basket of goods and services. Thus, PPPs are both price deflators and currency converters: they eliminate the differences in price levels between countries in the process of currency conversion.

There are various adjusted PPP conversion factors, which are appropriate for use in different contexts. PPP GNI per capita (formerly PPP GNP per capita) is used for conversions of WTP results in this report as it reflects differences in real income per capita and real income is a key determinant of WTP. The adjustment is a simple linear one. For example to convert a WTP estimate from the US to the UK, we undertake the following calculation:

$$\mathbf{WTP_{UK} = WTP_{US} \times (PPP\ GNI\ per\ capita_{UK} / PPP\ GNI\ per\ capita_{US})}$$

The other approach to spatial adjustment is to use the actual exchange rates, which is a cruder approach and is not used in this report.

Spatial adjustment is made in the year in which the original estimate is presented. Temporal conversions require choosing between price indices, given that price indices are disaggregated by commodity and service categories that better reflect the temporal change in relative prices for specific subgroups. WTP studies measure health benefits and environmental benefits and costs in terms of income and the associated consumption forgone. Accordingly WTP based estimates can be inflated or deflated by using the CPI. This is standard practice for adjusting WTP estimates in value across time (see Eisworth and Shaw 1997; Health Canada, Research Triangle Institute and USEPA, 2002). Again, this is a simple multiplication as shown below for an adjustment from 1998 to 2003:

$$\mathbf{WTP_{UK}03 = WTP_{UK}98 \times (CPI_{UK}02/CPI_{UK}98)}$$

For other economic values (not estimated through WTP) other indices are appropriate. For example, when adjusting changes in housing prices from hedonic studies, housing price indices are employed instead.

5. Conversion of kiel and mcclain results to £/tonne

Kiel and McClain employ hedonics to estimate changes to house prices that relate to proximity to the incinerator at North Andover, Massachusetts. The study dates from 1995, and in 1999 the USEPA fined the incinerator operators for violating the clean air act, in relation to mercury and lead emissions. This knowledge, combined with the fact that the study is from the US and the localised nature of the impact makes this case study a poor example for benefits transfer to the UK. Nonetheless, we reproduce below the calculations that were used to arrive at a per tonne of waste figure, that results from manipulation of the study results. There are a number of assumptions involved that were required due to lack of more specific data, e.g. in the form of detailed maps of the specific case study area.

Background data:

According to the United States Census Bureau, the town has a total area of 72.1 km² (27.8 mi²). As of the census of 2000, there are 27,202 people, 9,724 households, and 6,904 families residing in the town. The population density is 394.1/km² (1,020.7/mi²). There are 9,943 housing units at an average density of 144.1/km² (373.1/mi²).

In Massachusetts, nine incinerators burn 3.3millions tonnes of waste each year. Thus, the throughput of each incinerator is estimated to be 3.3million/9 = 366,667 tonnes per year.

From the study:

The study estimates that house prices increase with distance from the site of the incinerator at a rate of $6,607 per mile (in 1995 US$), up until 3.5 miles from the site, after which no discernable effect from the incinerator alone shows up in the analysis. Thus, by inference, house prices start declining at 3.5 miles from the site.

Calculation

In order to arrive at a £/tonne estimate, we need to calculate the total number of houses affected by the incinerator location. If we take three distance bands from the incinerator, and calculate the area of these distance bands at 1.16, 2.32 and 3.5 miles from the incinerator, then we can estimate the number of houses in these distance bands using the statistics on housing density. Then we can estimate the total depreciation on the stock of houses due to the disamenity of the incinerator with the following formula:

$$Total\_depreciation = (3 \times \$6{,}600 \times 373 \times \pi 1.16^2) + (2 \times \$6{,}607 \times 373 \times (\pi 2.32^2 - \pi 1.16^2))$$
$$+ (\$6{,}607 \times 373 \times (\pi 3.5^2 - \pi 2.32^2))$$

Thus, total depreciation is $146,426,984 (1995 prices). To arrive at an annual depreciation, we divide by the discount rate (3.5%), which gives us: $5,124,944 per year.

Dividing this figure by the total tons of waste processed at the facility (366,367 tons), we get: $14/ton (1995 prices).

Inflating this figure using the ratio of average house prices in Massachusetts in 1995 to the average house prices in 2003, then converting to £ using PPP GNI ratios, and rounding to two significant figures to reflect the uncertainty in the analysis, gives us £21/tonne of waste as the average disamenity value of an incinerator on the surrounding community.

6. Air pollution and ecosystem studies

Cost benefit analysis of proposals under the UNECE multi-pollutant, multi-effect Protocol (AEA Technology report for the Department of the Environment Transport and the Regions. 1998).

Study summary

This study estimates the pan-European benefits from reductions in air pollution emissions for two different scenarios (a baseline and a maximum feasible reduction scenario). The benefits include reduced impacts to human health, materials, crops, forests and visibility. However, the benefits analysis for the different receptors varies considerably in certainty. The authors of this report write that some of the benefits analyses push the quantification procedure to the extreme. This is particularly true for the assessment of damage avoided to ecosystems, forests and visibility. Consequently, they rank the benefits analyses into five groups. These range from group A, in which all respondents (i.e. study researchers and UK government officials) appeared to have confidence, to group E, in which no respondent had much confidence. A series of cost benefit comparisons are then made, whereby the costs of achieving emissions reduction are compared to the five different groups of benefit estimates.

Methodology for estimating damages to ecosystem from air pollution

AEA Technology (1998) bases the valuation of damage to ecosystems on one study. Navrud (1988) estimates the WTP to reduce SO_2 emissions in order to protect salmon and trout ecosystems in Norway. The study provides two WTP estimates, i) WTP of 59 Euro/hhld/year for a 30% reduction in SO_2 emissions and ii) WTP of 73 Euro/hhld/year for a 70% reduction in SO_2 emissions. AEA Technology (1998) assume a zero WTP for a 0% reduction in emissions. Using these three points they derive a simple relationship, given as:

WTP(Euro/hhld/year) = 17.2. ln(%SO_2 reduction)

Another study by Ecotec (1994) reports a WTP of 34 Euro/hhld/year for an 80% reduction in S emissions to upland ecosystems in the UK. The authors assume that the form of the relationship is the same simple logarithmic form inferred from Navrud's study thus another relationship is defined:

WTP(Euro/hhld/year) = 7.8. ln(%SO_2 reduction)

The authors themselves state that the 'use of these functions would clearly be subject to major uncertainty. Analysis based on their application is not suggested as ideal by any stretch of the imagination' (*sic*). They go on to list numerous necessary assumptions. The most significant of these are i) WTP estimates for the protection of fish in Norway and upland ecosystems in the UK can be reliably transferred to represent WTP for all other types of ecosystems in Europe; ii) a %

SO_2 reduction can be translated directly into % ecosystem protection. These assumptions are clearly very simplistic.

Unit damage values

Average unit damage values for SO_2, NO_x and NH_3 can be drawn from this study using total UNECE avoided damages divided by UNECE emissions reductions. However, due to the considerable uncertainty associated with the benefit values for ecosystems, forests and visibility, these values are generally omitted. This procedure was used for an earlier benefit assessment study which eftec contributed to for DG Environment: European Environmental Priorities: An Integrated Environmental and Economic Assessment (2000). This study made use of unit damage values which include only the most certain benefit assessments i.e. to health, materials and crops only.

It is also possible to dis-aggregate the average unit damage values to provide unit damage values for air borne pollutants to the individual receptors. However, for the same reasons given above we do not recommend taking such values from this study.

UNECE conference

Investigation of this conference as reported at the website given above revealed that economic valuation of pollution damage to ecosystems is still in its infancy. Papers presented at the conference include literature reviews and future research needs and other basic issues such as, 'can you use CVM to measure benefits in the context of ecosystems'?

In addition, investigation of the links to other sites, related to this field provide further evidence that this field of analysis is still in its early days and so far unable to provide unit damage values for air pollution to ecosystems.