Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes

Written by Enviros Consulting Ltd and University of Birmingham with Risk and Policy Analysts Ltd, Open University and Maggie Thurgood





Department for Environment, Food and Rural Affairs Nobel House 17 Smith Square London SW1P 3JR Telephone 020 7238 6000 Website: www.defra.gov.uk

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FOREWORD BY MINISTER OF STATE, ENVIRONMENT & AGRI-ENVIRONMENT

The Prime Minister's Strategy Unit, in its report "Waste not, Want not" recommended that an independent body should bring together the literature and evidence on the relative health and environmental effects of all the different waste management options; relative both to each other and to other activities affecting health and the environment. Defra commissioned this report in response to that recommendation.

The report examines the waste management options for treating municipal solid and similar waste. It focuses, as Defra requested, on the principal types of facilities that are currently used for dealing with such waste in the UK and in Europe and on what the currently available scientific evidence can tell us about their environmental and health effects.

It is a very comprehensive report and brings together, for the first time, a wealth of evidence which allows us to consider the health and environmental impacts of waste management on the basis of all available information.

The report has been peer reviewed by the Royal Society and I am grateful to Prof. Howard Dalton, Defra's Chief Scientific Advisor, for advising me on the scientific analyses.

I am particularly encouraged by the report's conclusion that, on the evidence from studies so far, the treatment of municipal solid waste has at most a minor effect on health in this country particularly when compared with other health risks associated with ordinary day to day living. The evidence on environmental effects is limited, but such as there is does not appear to suggest adverse environmental effects of waste management, other than those we know about and are already addressing, such as methane emissions from landfill.

The report rightly recognises that there is more that we can and should still learn and we will be addressing the need and priorities for further research through our waste research strategy this summer. The search for knowledge is never complete and this report usefully identifies areas of research that we will be taking forward as part of our continual efforts to refine the evidence base for policy making.

I believe that this report does give us sufficient confidence in our current policies for local authorities to press ahead urgently with the task of approving planning applications for new waste management facilities. Among the other conclusions to be drawn, the report shows that risks to human health from incineration are small in comparison with other known risks. We must acknowledge the role of incineration with energy recovery as a sustainable waste management option although the priority must be waste minimisation, reuse and recycling. Incineration is an option for dealing with the residual waste that will still be left even after achieving the much higher levels of recycling and reuse we are aiming for and to help absorb the diversion of municipal waste from landfill which we are required to make under the Landfill Directive.

FOREWORD BY MINISTER OF STATE, ENVIRONMENT & AGRI-ENVIRONMENT (CONTINUED)

We must manage the growing amount of waste we produce. We will do this by basing our policies on the best available scientific evidence and on an assessment of the comparative risks. We will continue to develop our scientific knowledge to support our policies. This report is a helpful contribution to that process.

Ellist Monley

Elliot Morley Minister of State for Environment & Agri-Environment

FOREWORD BY DEFRA CHIEF SCIENTIFIC ADVISER

Ministers asked for my assessment of this report in my role as Chief Scientific Adviser to the Department. This Foreword is the advice I have given in light of that request.

This is a timely and useful report, which for the first time provides Government with a critical assessment of the available peer-reviewed scientific literature on the health and environmental effects of options for managing municipal solid waste. I am grateful to the authors, Enviros Consulting Ltd., Professor Harrison, and their colleagues, for their comprehensive and thorough review, and for approaching a difficult task in a positive and imaginative way. I am also very appreciative of the work done by the Royal Society in providing a detailed critique which has been reflected in finalising this report. The Royal Society's working group provided valuable comments on the emerging report. Their statement of March 2004 reflects the extent to which their formal critique of the full draft of November 2003 has contributed in shaping the final version of the review. (These are reproduced at Appendix 4.)

The review and insights of the Royal Society's working group have been of great assistance in preparing my advice to Ministers on the science to support waste management policy. Particularly helpful in this regard is the critical assessment of the quality of the scientific evidence on each of the issues through the use of a 'reliability index', a feature that other similar assessments might adopt to advantage.

The review has concluded that the effects on health from emissions from incineration, largely to air, are likely to be small in relation to other known risks to health. I have confidence in this conclusion, particularly bearing in mind the fact that the current generation of municipal solid waste incinerators have to comply with much more stringent emission standards than those which formed the basis for the majority of studies of health effects in the literature. This does not mean that we can afford to be complacent; rigorous enforcement will be crucial to ensure that the new emission standards are not exceeded, and that non-standard operating conditions, as noted by the Royal Society, do not lead to levels of emission which would give rise to concern.

The review has also addressed the effects on the wider environment. The most important in this context is the contribution that landfill emissions make to emissions of methane, a powerful greenhouse gas. The review has also noted that odours from landfill can be important, and that measures to capture and use landfill gas could alleviate both of these potential problems. The review reported little existing evidence of other environmental effects due to waste management.

FOREWORD BY DEFRA CHIEF SCIENTIFIC ADVISER (CONTINUED)

The contributions of municipal solid waste to air emissions of methane (27% of UK total) and cadmium (about 10% of UK total) are well known to arise mostly from landfill. This is one of the reasons why government policy is moving away from the landfill waste option. With these exceptions, management of municipal solid waste accounts for less than 2.5% of all other emissions for which data are available (including carbon dioxide and toxic gases). These conclusions mean that the overall scale of direct effects of releases to air from waste management practices is relatively small compared with emissions from other sectors such as transport.

I am nevertheless aware that, since the review was first and foremost a review of the existing literature, coverage will be limited by the availability of evidence, so some areas of the science will be analysed in more depth than others. Consequently there will be gaps and uncertainties in the evidence base.

Areas where there is less work and the science is less certain include releases to soil and water and releases from composting, or other forms of waste management like mechanical biological treatment or anaerobic digestion. One other important study reported an association between birth defects and proximity to landfill sites. The authors of that study were clear, however, that the association reported in this single study does not demonstrate a causal relationship, and the current review reflects this. It would be desirable if further studies could be carried out to identify the non-waste related factors which may influence this association.

In order to reduce, or remove these uncertainties, and to fill gaps highlighted by the review as missing from the current literature, we will need to undertake further research. The issues suggested by the review will be included in consideration of priorities as the waste research strategy is developed, with interested stakeholders, and particularly with the Department of Health.

The Royal Society has highlighted the advantages offered by Life Cycle Analysis in extending the range and scope of comparative analyses available. Life Cycle Analysis (as advocated in Defra's Waste Strategy 2000, and used by the Environment Agency in their WISARD waste management software) is of particular relevance in recycling, and should be incorporated in future research design on this issue.

In conclusion, I welcome this report. Not collecting or managing waste is not an option. The formulation of policy on the management and disposal of waste is an important area of Defra's work. This report helps decision makers by bringing together and analysing the existing body of waste management research.

Anward Dalta.

Professor Howard Dalton FRS

1.	INTR	ODUCTION	1
	1.1	Background	2
	1.2	Sources and types of waste considered in the study	4
	1.3	Waste management facilities considered in this study	8
	1.4	Processes of Exposure	14
	1.5	This report	19
	Anne	ex 1.1 : Specification	21
2.	REVI	EW OF INFORMATION ON EMISSIONS	23
	Sumr	mary –information on emmissions	24
	2.1	Introduction	25
	2.2	Scope of the Report	26
	2.3	Sources of Information	30
	2.4	Materials Recycling Facilities	31
	2.5	Composting (In-vessel)	34
	2.6	Composting (Windrow)	41
	2.7	Mechanical Biological Treatment (MBT)	47
	2.8	Anaerobic Digestion with Energy Recovery	51
	2.9	Gasification/Pyrolysis with Energy Recovery	57
	2.10	Unsegregated Incineration with Energy Recovery	64
	2.11	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	82
	2.12	Landfill with Landfill Gas Flaring and/or Energy Recovery	85
	2.13	Waste Transportation excluding Collection	113
	2.14	Summary	114
3.	REVI	EW OF EPIDEMIOLOGICAL RESEARCH	119
	Sumr	mary – epidemiological research	120
	3.1	Glossary	121
	3.2	Introduction	125
	3.3	Materials Recycling Facilities	130
	3.4	Composting	133
	3.5	Other Waste Disposal Options	138
	3.6	Incineration	139
	3.7	Landfill	159
	3.8	Tables	164

4	οuΔ	NTIFIC	ATION OF THE HEALTH CONSEQUENCES OF EMISSIONS	190
	Sumi	marv –	quantification of health consequences	101
	⊿ 1	Scone		102
	4.1	Metho	d for assessing the health effects of emissions to air	192
	л. <u>2</u> Л.3	Proces	sees and Substances Considered	104
	4.5	Proce		100
	4.4	Diepor	reion modelling	202
	4.5	Posuli		202
	4.0	Comp	ariaan of baalth offacts of omissions to air from wasto	204
	4.7	manag	gement facilities	207
5.	QUA EMIS	NTIFIC	ATION OF THE ENVIRONMENTAL CONSEQUENCES OF	213
	Sumi	mary –	environmental consequences of emissions	214
	5.1	Introd	uction	215
	5.2	Sourc	es of Information	216
	5.3	Main I	Findings	218
	5.4	Shorto	comings in information	223
6.	CON REVI	TEXT F	FOR QUANTIFIED HEALTH AND ENVIRONMENTAL RISKS, AND F PUBLIC PERCEPTION ISSUES	224
	Sumi	mary –	contextual information and public perception	225
	6.1	Introd	uction	226
	6.2	Comp	arison between MSW management options	228
	6.3	Comp	arison between MSW management and other activities	234
	Anne	x 6.1 :	Public perception of waste management	245
7.	CON	CLUSI	ONS	248
	Sumi	mary		249
	7.1	Introd	uction	250
	7.2	Emiss	ions	252
	7.3	Health	n Effects	259
	7.4	Enviro	onmental Effects	262
	7.5	Recon	nmendations for future work	263
APPI	ENDIX	(1:	BIBLIOGRAPHY	
APPI	ENDIX	(2:	LITERATURE REVIEW OF THE ENVIRONMENTAL EFFECTS OF WASTE MANAGEMENT OPERATIONS	
APPI	ENDIX	(3:	ELECTRICITY GENERATED FROM MSW ENERGY RECOVERY	
APPI		(4:	ROYAL SOCIETY STATEMENT	

Introduction	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		antification of the vironmental nsequences of issions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Background		Sources ar	nd types of waste		Waste management facilities	;	Health eff	ects		This report

1. INTRODUCTION

Introduction	Review of Information on Emissions		Review of Epidemiological Research	Quantification of Health Consequences of Emissions		antification of the vironmental nsequences of issions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Background		Sources ar	nd types of waste	Waste management facilities		Health eff	ects		This report

1.1 Background

Enviros Consulting Ltd and Birmingham University were invited by DEFRA to carry out a review of the health and environmental impacts of waste management.

The commissioning of this study arose from a recommendation in the Strategy Unit report "Waste not, Want not: A strategy for tackling the waste problem in England," and a subsequent commitment in the Pre-Budget Report 2003. Recommendation 15 of the Strategy Unit report states that:

"An independent body should bring together the literature and evidence on the relative health and environmental effects of all the different waste management options; relative both to each other and to other activities affecting health and the environment."

The commitment in the Pre-Budget Report states,

"The Government will ... commission a review of the environmental and health effects of all waste management and disposal options. The case for using economic instruments for incineration will be considered in light of this work, and in consultation with other stakeholders. The Government is also considering how the use of economic instruments could be extended further."

DEFRA has also commissioned a separate study to examine the range of economic values for the impacts and pollutants identified in this study. Enviros Consulting Ltd, in association with Eftec Ltd are undertaking this work

Readership

The report is designed to be suitable for use in informing and supporting waste management decisions. This report is also intended to be suitable for use as background information to assist in consideration of the case for using economic instruments for incineration. A separate report has been prepared by Enviros Consulting Ltd and Eftec for DEFRA, which considers the economic costs and benefits of the health and environmental effects of waste management.

The full report is accompanied by an extended summary document, setting out the key findings of the research.

This study reviews the health and environmental effects which can be linked directly to facilities treating MSW and similar wastes. To fully support waste management decisions, this report needs to be read in conjunction with other information relating to waste management. Some points of reference which may be helpful are set out below; a full bibliography is given in Appendix 1.

- Life-cycle effects of waste management: A series of reports have been published by the Environment Agency, developing life cycle inventories for composting, anaerobic digestion, incineration, landfill, recycling and transportation of waste (see Environment Agency 2000a, 2000b, 2000d, 2000f, 2001c). There are a number of tools available for investigating the life-cycle effects of waste management options, including the Environment Agency's WISARD model (see <u>www.environment-agency.gov.uk/subjects/waste</u>)
- Economic effects of waste management: The report prepared for DEFRA by Enviros Consulting Ltd and Eftec reviews the economic effects of waste



Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qui Env Cor Em	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities	5	Health eff	iects		This report

management. A substantial database of information on the economic costs of a range of environmental emissions and impacts has been prepared under the European Commission's ExternE programme (see http://externe.jrc.es)

- Local environmental information: When considering the environmental effects of particular waste management operations, it is important to consider the local environmental conditions.
 - Information on designated habitat sites and other land designations in England can be obtained from the Multi-Agency Geographic Information for the Countryside (<u>www.magic.gov.uk</u>). Information for other parts of the UK is available off-line from Scottish Natural Heritage, the Countryside Council for Wales, and the Northern Ireland Environment and Heritage Service.
 - The Environment Agency provides a series of databases via its website entitled "What's in your backyard?" These provide information on bathing water quality, discharges to sea, river water quality and water management, and the presence of landfill sites and processes regulated by the Environment Agency
 - Information on air quality across the UK can be obtained from the air quality archive (<u>www.airquality.co.uk</u>). This archive contains an extensive database of air monitoring information, together with estimated levels of certain key pollutants across the UK.
 - Local authorities should be consulted for other information on local environmental conditions
- Emissions from particular facilities: information on emissions from existing industrial processes regulated by the Environment Agency is available through the Environment Agency "What's in your backyard?" database. Information on proposed processes or processes operating elsewhere can be obtained by discussion with the manufacturer. If a planning application or pollution control permit application has been submitted for a specific process, this should include information about emissions and an assessment of potential environmental effects.

Municipal waste facilities

This report addresses the impacts of the range of facilities commonly used to handle municipal waste and similar wastes. This includes household waste and similar commercial, industrial and institutional wastes – for example, wastes from shops, offices, restaurants and hotels. A second report will address the potential effects of facilities handling other waste streams.

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Issue	antified Risks, & lic Jes	Conclusions
Background		Sources an	nd types of waste	Waste management facilities		Health eff	ects		This report

1.2 Sources and types of waste considered in the study

This report is concerned with environmental and health impacts of the principal facilities for disposal of MSW (municipal solid waste) and similar waste streams. Other types of waste (such as industrial, agricultural, hazardous or mineral wastes) are specifically excluded from consideration and will be the subject of a later report. However, much of the commercial and some of the industrial waste generated in the UK is similar in nature to municipal solid waste. The conclusions of this report will therefore be applicable to these other waste streams. This is particularly the case as regards landfill, because MSW is often landfilled along with other waste streams. This means that the data on emissions, health effects and environmental effects associated with landfill is actually obtained from landfills accepting a wider range of waste materials.

In Britain, municipal waste is defined as waste collected by, or on behalf of local authorities and includes the following waste types:

- Household wastes (collected waste, waste collected for recycling and composting and waste deposited by householders at household waste disposal sites); this accounts for 89% of MSW;
- Household hazardous wastes;
- Bulky wastes derived from households;
- Street sweeping and litter;
- Parks and garden wastes;
- Non-hazardous trade wastes collected by local authorities (this component tends to be limited; Strange, 2002);
- Wastes from institutions, such as schools, etc.

In the UK, taking into account all waste arisings, 430 million tonnes of waste was generated in 2000/01. Approximately 7% of this (28.8 million tonnes) was MSW (Office of National Statistics, 2003; DEFRA 2003).

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research		Quantification of Health Consequences of Emissions	Qua Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic Jes	Conclusions
Background		Sources an	id types of waste	,	Waste management facilities		Health eff	ects		This report

Figure 1.1 Waste production in the UK, 2000/01



Municipal solid waste consists of many different materials. Its composition is variable from place to place and time to time, depending on a range of factors. Household waste reflects population density and economic prosperity, seasonality, housing standards and presence of waste minimisation initiatives (for example home composting). The make-up of commercial waste will be influenced by the nature of commerce in a local area (Strange, 2002). The most common components of MSW are: paper and cardboard (33%); garden waste and other organic matter (21%); plastics (13%); food wastes (11%) (vegetable and animal matter, oils and fats) ; fabrics (5%); glass (4%); metals from containers and packaging (4%);. Figure 1.2 provides an illustrative breakdown of materials in MSW collected from domestic dwellings (House of Commons, 2001).

Introduction	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		antification of the vironmental nsequences of issions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Background		Sources an	nd types of waste		Waste management facilities	5	Health eff	ects		This report

Figure 1.2 Typical breakdown of materials in MSW by weight



The resource cycle

As with all waste materials, MSW forms part of a resource cycle. The production of waste is related to the use and consumption of materials by all sectors of society. The production of waste within a cycle of resource use, processing, re-use and disposal is illustrated in Figure 1.3.

Introduction	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		antification of the vironmental nsequences of iissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Background		Sources an	d types of waste		Waste management facilities	5	Health eff	ects		This report

Figure 1.3 Illustration of the resource cycle



Courtesy of Professor William Powrie, University of Southampton

As set out in Annex 1.1, this project considers the aspects of the resource cycle directly relating to the treatment and disposal of municipal solid waste. Other components of the resource cycle such as extraction of raw materials, transportation of raw materials and goods, and manufacture of products would be relevant if considering the life-cycle impacts of particular material flows. Direct emissions from transportation and reported environmental effects of transport of MSW are addressed in this project, while indirect effects of transportation lie outside the scope of this report.

The collection, treatment and disposal of MSW and similar wastes provides a benefit to communities by removing wastes which would otherwise decompose and become a source of disease, pests and nuisance.

As shown in Figure 1.3, one key offset from waste management is the production of energy from some waste management facilities. These include anaerobic digestion facilities, pyrolysis/gasification facilities, waste-to-energy incinerators, and landfills with generating engines. Occasionally, waste management facilities generate heat in the form of steam or hot water (one example of this is the incinerator at Nottingham which provides heat for housing, industry and public buildings). More commonly, the waste is used to generate electricity which can be exported to the national grid. This can result in an offset in emissions, as a result of a reduced need to generate electricity from other sources such as fossil fuels. Any reduction in emissions would occur at a different location to the waste management facility. The energy generated from different waste management activities is set out in Appendix 3. The net effect of this on emissions is discussed briefly in Chapter 2.



Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources an	nd types of waste	Waste management facilities	5	Health eff	ects		This report

1.3 Waste management facilities considered in this study

The following facilities or stages in the management of municipal solid waste are in common use:

- Minimisation of waste materials (e.g. via purchasing policy; repair in preference to replacement; home composting)
- Product re-use. Re-using products may result in an overall offset in environmental effects due to a lower requirement for manufacture of replacement products, although we have not assessed any such benefits in this report.
- Material recycling
 - This comprises the separation, treatment and re-use of waste as a secondary raw material.
 - This includes composting of green wastes
 - Handling facilities for recycling of MSW will include collection, transfer stations/materials recycling facilities (MRFs), composting facilities, and mechanical-biological treatment (MBT) facilities
 - Recycling waste materials in this way may result in an overall offset in environmental effects due to a lower requirement for raw materials. Conversely, the recycling of materials removed from MSW is likely to result in environmental impacts associated with the reprocessing operation. The potential environmental effects other than those associated with facilities handling MSW have not been assessed in detail in this report, although we have highlighted where they may need to be considered.
- Thermal treatment
 - Treatment of the waste usually with generation and recovery of thermal and/or electrical energy. The generation of electricity in this way results in a reduction in the need to generate energy in other ways, and hence an offset to any adverse environmental effects of electricity generation.
 - Facilities for thermal treatment include anaerobic digestion, gasification/pyrolysis and incineration
- Landfill of residual materials

These are not direct alternatives to one another since only landfill and incineration are capable of dealing with the entire mass of MSW without prior sorting, and landfill of some residues is virtually inevitable. However, landfill and/or incineration might form a component of a waste management strategy which also provides for substantial recycling and composting.

Figure 1.4 illustrates the proportion of MSW disposed of by each primary route in England in 2001/02.

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	id types of waste	Waste management facilities	5	Health eff	iects		This report

Figure 1.4 Municipal waste management in England, 2001/02



Future trends in management of MSW

As shown in Figure 1.4, landfill is currently by far the most common MSW disposal route. This is likely to change in the future. The Landfill Directive (1999/31/EC) aims to prevent, or to reduce as far as possible, the negative environmental effects of landfill, and will require *inter alia* a progressive reduction in the weight of biodegradable municipal waste sent to landfill up to 2020:

- Reduce BMW landfilled to 75% of 1995 level by 2010;
- Reduce BMW landfilled to 50% of 1995 level by 2013;
- Reduce BMW landfilled to 35% of 1995 level by 2020.

The Government's Waste Strategy 2000 (DEFRA, 2000) also sets targets for progressive increase in recycling up to 2015.

MSW management methods covered in this study

Links between different MSW management methods are illustrated in Figure 1.5 below.

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qui Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	d types of waste	Waste management facilities	5	Health eff	ects		This report

Figure 1.5 Material flows within the management of MSW



STW: Sewage treatment works

Materials Recycling Facility

A materials recycling facility (also known as a materials recovery facility or materials recycling factory) is defined as a central operation where source segregated, dry recyclable materials are sorted mechanically or manually to market specifications for processing into secondary materials (Gladding, 2002). As implied by this definition, waste material entering a materials recycling facility has normally been subject to some pre-segregation by the householder, but further mechanical or manual sorting is required. MRFs are widely used for waste pre-sorting in the UK.

As noted above, recycling waste materials in this way may result in an overall offset in environmental effects due to a lower requirement for raw materials. Waste materials which can be recycled include paper, metals, glass and plastics. In the case of glass recycling, for example, the use of glass cullet improves furnace efficiency compared to the use of raw materials alone. Conversely, the recycling of materials removed from MSW is likely to result in environmental impacts associated with the reprocessing operation and transportation of materials to the reprocessing facility. As set out in Annex 1.1, we have not assessed the environmental effects of these secondary activities, focusing on those directly associated with facilities for sorting MSW.

Composting

The composting process is the controlled biological decomposition and stabilisation of organic materials, such as vegetable, plant and some food wastes, in the



Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities		Health eff	ects		This report

presence of air, resulting in biologically produced heat. It results in a final product typically referred to as "compost" which has been sanitised and stabilised, is high in humic substances and, if free from contamination, can be beneficially applied to land. Composting is now employed as a treatment process for a wide range of organic substrates such as municipal solid wastes, sewage sludges and agricultural and industrial bio-products.

Composting of unsorted MSW is not widespread in the UK, although composting of the green waste fraction of MSW is widely practised.

Mechanical Biological Treatment

Mechanical Biological Treatment (or whole waste composting) (MBT) is a generic term for an integration of several processes. MBT processes are usually designed to stabilise the residual waste stream after initial recyclables and compostables are removed or prepare it as a fuel for a thermal treatment process. The process may involve separation of the non compostable element of the waste stream with screening of the output material into a reject fraction. The remaining component may be composted or fed into an anaerobic digester. Further screening or sorting may be required, dependent on the ultimate application of the residue.

MBT is not widely used in the UK at present.

Anaerobic digestion

Anaerobic digestion is a biological process which produces biogas from organic materials such as the organic component of MSW. Biogas comprises methane and carbon dioxide. The process takes place in the absence of oxygen in an airtight container known as a "digester." Decomposition in the absence of oxygen produces biogas, containing mainly methane and carbon dioxide. The biogas produced from anaerobic digestion is normally burnt to provide heat and/or electricity. If the digested material is of suitable quality, it can be spread to land, improving land quality and reducing the need for chemical fertilisers.

Anaerobic digestion is applied to agricultural wastes and sewage sludge in the UK, but not yet widely used on MSW.

Pyrolysis/gasification

As the name suggests, this is a multi-stage process. In the pyrolysis stage, the waste materials are heated in the absence of oxygen. Organic materials are converted to simple gases leaving a residue of carbon char which contains inert materials and any heavy metals. In the gasification stage, carbon residues are reacted out with air and steam in the "water-gas" reaction to produce hydrogen and carbon monoxide. Finally, these gases are combusted to produce energy and heat.

Pyrolysis/gasification of MSW is operated at pilot scale in the UK at present.

Incineration

Incineration involves combustion of waste at high temperatures for a sustained period achieving a very substantial reduction in the volume of waste and effectively destroying pathogenic biological organisms. The by-products of the combustion process comprise principally emissions to atmosphere and residual ash. These are quantified in Chapter 2 of this report.

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qui Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Issue	antified Risks, & lic Jes	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities	5	Health eff	ects		This report

Incineration of MSW is an established technology. In England and Wales, there are 14 operational MSW incinerators, dealing with 9% of MSW generated. Regulation of MSW incinerators has become increasingly stringent over the past ten years, following the implementation of various European directives, most recently the Directive on the Incineration of Waste (2000/76/EC).

Landfill

The practice of landfill involves the creation of contained void spaces. These are normally in the form of cells, filled with compacted waste materials which are progressively covered, then sealed with a permanent cap. Since much of the waste is not processed prior to disposal in a landfill, biodegradable materials subsequently decay releasing landfill gas. Landfill gas comprises mainly methane and carbon dioxide, and is increasingly collected for combustion and energy conversion. The same decomposition, and the passage of water through the waste, give rise to leachate, a contaminated liquid that is collected, removed and treated.

Landfill design and operation has improved continuously and substantially since the early 1970s to the present day, where major changes are currently taking place with the implementation of the Landfill Directive and Integrated Pollution Prevention and Control.

Landfill differs from the other waste management facilities considered in this report, in terms of the timescale over which materials disposed to landfill could potentially affect human health or the environment. The majority of emissions of landfill gas and leachate from biodegradable waste materials typically take place over a period of some 20 years following disposal. However, generation of gas and leachate will continue at a lower rate for many years.

Transportation

In addition to the impact of releases associated with these processes, an assessment has been made of the impacts of transport activities associated with MSW disposal.

Operation and regulation of MSW facilities

Local authorities are responsible for the collection and disposal of MSW. In two-tier authority areas, the borough or district council is responsible for collection of MSW, and the County Council is responsible for the disposal of MSW. In single authority areas, the unitary authority is responsible for both collection and disposal of MSW.

Facilities for handling and disposing of MSW are usually operated by commercial organisations under contract to the County Council or unitary authority. Operators of waste management facilities are required to have a valid planning permission for the activity(ies) being undertaken.

Together with agricultural wastes and commercial/industrial wastes, MSW is a Controlled Waste. This means that facilities handling MSW need to hold a Waste Management licence, issued by the Environment Agency under the Waste Management Licensing Regulations 1994 (Statutory Instrument 1994 No. 1056). For some operations, licensing under the Waste Management Licensing Regulations is being progressively replaced by permitting under the Integrated Pollution Prevention and Control regime.

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qui Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities	5	Health eff	ects		This report

Non-standard operating conditions

This project focuses principally on the effects of operation of waste management facilities in accordance with regulatory requirements. Occasionally, however, any industrial facility will experience incidents when operation does not take place normally, and regulatory requirements are not met. Examples of this might include an interruption to the waste feed into a pyrolysis/gasification plant; a change in the composition of materials fed to an incinerator resulting from inadequate mixing of wastes; or a temporary interruption to the operation of a landfill gas flare.

Since the epidemiological evidence for human health effects and field evidence for environmental effects are obtained from studies of facilities operating in the real world, they are likely to span periods of time in which MSW management installations were operating under abnormal conditions. This means that the epidemiological evidence for the presence or absence of health and environmental effects of MSW management provided in Chapter 3 and Chapter 5 include any effects of operation under abnormal operating conditions.

Information on emissions in Chapter 2, on which the assessment in Chapter 4 is based, is based largely on measurements and estimates representing normal operation. For short periods, it is possible that emissions and health effects could exceed those presented in Chapter 2, and used as the basis for quantification of effects in Chapter 4. This is likely to be less relevant for health effects such as cancer, which are usually regarded as being dependent on prolonged exposure. Long-term exposure to emissions from waste management facilities will not normally be significantly affected by short-term abnormal operating conditions. Emissions under abnormal conditions might be more relevant for other endpoints such as respiratory irritation and compounds with developmental toxicity. In Chapter 2, we discuss in outline the kind of abnormal operating conditions which might occur, and how these might affect emissions. We also discuss the history of operation of UK landfills and incinerators out of compliance with operating licences and permits.

A history of operating outside regulatory requirements is not necessarily a guide to future practice. For example, landfill design and regulation have been substantially improved over the past 25 years. Similarly, the number of breaches of MSW incinerator licence conditions is decreasing from year to year.

Uncertainty

This report considers the uncertainty and reliability of the information provided, to enable the reader to understand the confidence that can be placed in the information. This also enables a view to be taken as to areas where further work could usefully be focused.

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qu Env Coi Em	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Issue	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities		Health eff	ects		This report

1.4 Health effects

The focus of this project is to improve our understanding of emissions from operations involving MSW, and our understanding of the health impacts of managing MSW. The information in this report could be used to support a "source – pathway – receptor" model for risk assessment of an individual facility, or of a waste management strategy (see, for example, DETR/Environment Agency, 2000). This model is often used to assess the health and environmental risks of waste management activities or facilities. This report can be used to provide information on how a particular source-pathway-receptor linkage interacts with the potential for risk to health or the environment. The research set out in Chapter 3 and Chapter 5 also indicates whether there is empirical evidence for theoretical health and environmental risks being realised in practice.

Processes of exposure

Possible sources, pathways and receptors associated with management of MSW are set out in Table 1.1. Table 1.1 refers to "potential" effects, as any emissions from individual facilities will normally be controlled to an acceptable level via appropriate design, operation and regulation.



Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qu En Co En	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities	5	Health eff	iects		This report

Table 1.1Sources, pathways, emissions and potential effects of waste managementmethods

Source	Emission(s) Pathway(s) Receptor(s)		Receptor(s)	Potential effects	
(Waste disposal method)				Human	Environmental
Landfill	Dust; odour; micro- organisms; litter; landfill gas (CH ₄ , CO ₂ and numerous trace compounds); exhaust gases from combustion of landfill gas (including carbon dioxide, carbon monoxide, oxides of nitrogen, sulphur dioxide, and other trace components)	Air- emissions of materials to air directly from the landfill during tipping, compacting, covering and storage activities; emissions to air of fugitive landfill gas; emissions to air of products of landfill gas combustion.	Nearby sensitive receptors in the vicinity of the landfill site; nearby sensitive habitats	Potential for exposure to a variety of potentially harmful materials which have been investigated in connection with birth defects, asthma, respiratory disease and cancer	Potential for soil acidification due to deposition of acid gases; increases in soil metals; vegetation damage due to oxides of nitrogen (NO _x) and sulphur dioxide (SO ₂)
	Leachate containing salts, heavy metals, biodegradable and persistent organics to groundwater, surface water and sewer	Water- leaching of materials into groundwater and surface waters due to fugitive escapes of leachate; emissions of treated and untreated leachate via permitted routes	Nearby sensitive receptors, groundwater users and surface water users; nearby sensitive habitats		Potential for contamination of ground and surface water with metals, organic compounds, bioaccumulation of toxic materials
	Metals (Zinc (Zn), lead (Pb), copper (Cu), arsenic (As)), and various organic compounds	Land- contamination of land during post- operative phase	Nearby sensitive receptors and users of post operative site		Potential for contamination of flora and fauna in contact with contaminated land, and possible bioaccumulation of toxic materials in flora and fauna
Thermal treatment (including inciner- ation)	Emissions of SO ₂ , NO _x , hydrogen chloride, hydrogen fluoride, volatile organic compounds (VOCs), carbon monoxide, carbon dioxide (CO ₂) nitrous oxide (N ₂ O), dioxins and furans, metals (Zn, Pb, Cu, As), dust, odour, micro-organisms	Air- emissions from waste during handling and storage operations; emissions of materials during handling of waste ash; emissions of gases and particles from combustion of waste	Nearby sensitive receptors; nearby sensitive habitats; sensitive receptors within the influence radius of the combustion gas plume; sensitive receptors exposed to ash during re-use	Potential for exposure to harmful materials which have been investigated in connection with cancer, asthma, respiratory disease, birth defects	Potential for soil acidification due to deposition of acid gases; increases in soil metals/dioxins; vegetation damage due to NO _x and SO ₂
	of combustion	of hazardous	aquatic habitats;	effects likely	contribution to



Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qu En Co En	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities	6	Health eff	iects		This report

Source	Emission(s)	Pathway(s)	Receptor(s)	Potential effects	
(Waste disposal method)				Human	Environmental
	sulphuric, carbonic and nitric acids, particulate matter, metals (including Zn, Pb, Cu, As), dioxins and furans	surface water; In some cases discharge of waste coolant water to licensed discharge point	downstream of waste water treatment works; receptors downstream of final waste water sludge effluent disposal route		acidification
	From ash: metals (including Zn, Pb, Cu, As), dioxins and furans; From deposition of combustion gases: sulphuric, carbonic and nitric acids, particulate matter, metals (including Zn, Pb, Cu, As), fluoride, chloride, dioxins and furans	Land- disposal of bottom ash and fly ash residues to land via ash reuse programs; leaching of materials from landfilled ash; deposition of combustion gases and particles to land from airborne emissions	Sensitive receptors exposed to soil contaminated with ash or deposited emissions, or to produce grown in contaminated soil;	Potential exposure to metals, dioxins and furans. Has been investigated in relation to cancer and birth defects.	No significant effects likely
Com- posting	Methane, carbon dioxide, dust, odour, bacteria, fungi	Air- emissions of from waste handling, compost generation and compost removal operations	Nearby sensitive receptors	Potential for exposure to harmful bacteria and fungi. Investigated in connection with respiratory and other diseases	No significant effects likely
	Trace contaminants in original compost feedstock. Might include: metals and organic compounds	Land- potential for transfer of contaminants from compost into subsequently treated soils, and potential for contamination of food chain	Sensitive receptors exposed to soil fertilised with compost and to produce grown in contaminated soil	Potential for exposure to contaminants in original feedstocks via deposition to soils when compost used on soils. Potential for uptake by produce of fertilised land	Potential for increase in contaminants in original feedstocks when compost used on soils.
Materials Recycling Facility	Dust and odour	Air- emission of materials during waste storage and sorting	Nearby sensitive receptors	Potential for dust and odour nuisance; possible ill health due to dust inhalation	No significant effects likely
	Organic compounds, produce residues, surfactants	Water- emissions of materials during cleaning of facility and materials,	Receptors downstream of waste water treatment works; receptors downstream of final sludge effluent disposal route	No significant effects likely	No significant effects likely

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qu Env Coi Em	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources an	nd types of waste	Waste management facilities	5	Health ef	fects		This report

Source	Emission(s)	Pathway(s)	Receptor(s)	Potential effects	
(Waste disposal method)				Human	Environmental
	Non-recyclable materials from feedstock	Land - emissions arising from landfilling of final residues	Receptors in vicinity of landfill used to dispose of final residues	No significant effects likely	No significant effects likely
Transport- ation	Vehicle emissions (including: carbon monoxide, carbon dioxide, nitrogen oxides, particulate matter, metals, rubber dust, VOCs. From accidental spillages: VOCs, dust, odour, litter	Air- emissions associated with vehicle operations; emissions from accidental spillages	General public, sensitive receptors in the vicinity of transfer stations or the final reception point	Potential for exposure to exhaust fumes along transport routes and at transfer stations.	Potential for exposure to exhaust fumes along transport routes and at transfer stations.
	Fuel derived VOCs, (diesel and petrol); surfactants and liquid wastes from cleaning	Water- potential for contamination of groundwater and surface water arising from accidental spills of waste water and during cleaning processes	Sensitive receptors in the vicinity of transfer stations or the final reception point	Potential for contamination of groundwater used as water supply, and potential contamination and subsequent exposure to surface waters	Potential for contamination of groundwater or surface waters

Toxicity

An individual might be exposed to a particular substance by one or more routes – for example, by breathing in air containing the substance; by consuming food or drink containing the substance; or by contact of the substance on the skin. Any substance to which we are exposed has the potential to cause harmful effects. The harm that might be caused by exposure to a particular substance is determined by the dose – that is, the amount of substance experienced by an individual. In general, the higher the dose, the higher the risk of adverse effects, and the more severe any effects would be expected to be.

This means that at a high enough dose, even an innocuous substance such as water can be lethal. Conversely, at low enough doses, no substance will be toxic (with the exception of a few cancer-causing chemicals, and even for these substances, exposure at very low concentrations will have a vanishingly small likelihood of any significant effect on health). Even substances that are essential to our bodies, such as iron, can be toxic at high doses.

In common with all other materials, the substances covered in this report have the potential for adverse health effects, if human exposure were to occur at a sufficient level. An introduction to the toxicity of chemicals commonly found in the environment can be found in the World Health Organisation/International Labour Organisation/United Nations Environment Programme document "Hazardous chemicals in human and environmental health," available online at http://www.who.int/pcs/training_material/hazardous_chemicals/section_3.html.

The substances emitted from management of MSW have a range of potential health effects, dependent on the dose received. The epidemiological research set out in



Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qui Env Cor Em	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities	5	Health eff	ects		This report

Chapter 3 indicates that adverse health effects are in general not observed in populations living close to MSW management facilities. This means that exposure to the substances emitted from these facilities is too low for significant adverse health effects to be observed in the field. However, an indication of the type of health issues which might in principle be of concern to researchers investigating the potential health effects of MSW management can be gained from considering the health effects which could arise from a sufficiently high dose of the substances emitted from these facilities:

- Eye irritation: volatile organic compounds
- Bronchitis: particulate matter, sulphur dioxide
- Increased susceptibility to respiratory infection: sulphur dioxide
- Asthma attacks: nitrogen dioxide
- Reduction in oxygen-carrying capacity of blood: carbon monoxide
- Effects on the central nervous system: lead, manganese, carbon monoxide
- Effects on the immune system: lead, dioxins, mercury, polycyclic aromatic hydrocarbons, benzene, polychlorinated biphenyls, organochlorine compounds including vinyl chloride, nickel, chromium, toluene
- Reproductive effects: arsenic, benzene, cadmium, chlorinated compounds, lead, mercury, polycyclic aromatic hydrocarbons, polychlorinated biphenyls
- Cancer: polycyclic aromatic hydrocarbons, arsenic, nickel, chromium, vinyl chloride, benzene
- Effects on the liver: arsenic, polychlorinated biphenyls, chloroform, vinyl chloride
- Effects on the kidney: mercury, cadmium, chromium, arsenic, lead, halogenated hydrocarbons, organic solvents, and pesticides

As noted above, these health effects would only be expected to arise if exposure was sufficient. The evidence set out in Chapter 3 of this report suggests that in exposure at sufficient levels does not normally arise.

The majority of these health effects are characterised by a threshold of effects – that is, a level of exposure below which no adverse health effects would be expected. The severity of any effect is likely to increase as the dose increases. This is sometimes referred to as a "non-stochastic" effect. However, in the case of some carcinogens, there is in principle no reason to expect there to be a threshold of effect – this is referred to as a "stochastic" effect. In this case, the severity of the effect is not related to the dose, but the likelihood of the effect occurring is related to the dose.

In common with other industrial processes, waste management facilities are operated and regulated to ensure that releases of potentially harmful substances are minimised and controlled to a level which is not significant. Chapter 2 sets out the residual emissions of substances of concern, and the health effects which are observed as a result of these emissions are set out in Chapter 3.

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Env Cor Em	antification of the vironmental nsequences of issions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources ar	nd types of waste	Waste management facilities	6	Health eff	ects		This report

1.5 This report

Literature review

The main purpose of this project was a review of published literature and other relevant information:

- A review of published literature on the health and environmental effects of MSW management. A total of 102 papers were used in this review.
- A compilation of information on emissions from MSW management facilities to air, land, groundwater, surface water and sewer. This information was provided directly by operators; taken from other research papers; and drawn from the Environment Agency's Pollution Inventory. A total of 520 papers and other sources of information were reviewed during the course of this research.

Projection of reported emissions

The project also developed the information on emissions by investigating the likely health effects of emissions to air from MSW management facilities. This extended the findings of a research project being carried out for the Environment Agency.

Report structure

The report is structured as follows:

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qu Env Coi Em	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background		Sources a	nd types of waste	Waste management facilities	5	Health eff	iects		This report

Chapter 2: Review of research and information on emissions from MSW management facilities

Emissions to air, land, groundwater, surface water and sewer were considered. This review focused on characterising emissions of various substances per tonne of waste processed in different facilities. The uncertainty associated with emissions information was evaluated. Based on the information for individual facilities, where possible a national total of emissions from waste management was estimated.

Chapter 3: Review of epidemiological research into the health effects of MSW management facilities

The evidence in Chapter 2 sets out information on the emissions from MSW management facilities which have the potential to affect health. Chapter 3 sets out evidence of whether adverse health effects arising from waste management operations are observed in practice. Where possible, dose-response functions were developed to help in characterising the health burden of emissions to air from waste management operations.

Chapter 4: Quantification of the health effects of MSW management facilities

Drawing on the information in Chapter 2, this chapter describes the use of an air modelling technique to estimate public exposure to levels of air pollutants around waste management facilities. Based on this, the health effects of exposure to these air pollutants could be quantified, and compared with the observations in Chapter 3.

Chapter 5: Review of research into environmental effects of MSW management facilities

The evidence in Chapter 2 sets out information on the emissions from MSW management facilities which have the potential to affect the environment. Chapter 5 sets out evidence of whether adverse environmental effects arising from waste management operations are observed in practice.

Chapter 6 : Contextual information

This information sets the effects of waste management in context with other familiar environmental and health issues, and draws conclusions supported by evidence. It will be of interest to all stakeholders in the waste management process

Chapter 7 : Conclusions

Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qu Env Coi Em	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions
Background Specification		ecification	Sources and types of waste		Health eff	iects	Pro	ocesses of exposure	

Annex 1.1 : Specification

The specification provided by DEFRA on 25 March 2003 extended the Strategy Unit and Pre-Budget Report commitments to clarify that this review of health and environmental effects should:

i. Collate the available information on the impacts on human health and the environment from different options for the management and disposal routes of waste and to evaluate this information in terms of its scientific robustness

In these terms of reference, environment covers both the natural environment, such as for example, impacts on flora and fauna, water systems, odour, noise levels and the man-made environment, such as for example buildings and materials

- ii. Compare the performance of waste management and waste disposal options relative to one another across all categories of environmental and health impacts;
- iii. Compare the performance of waste management and waste disposal options relative to "other polluting activities" affecting human health and the environment ("other polluting activities" are taken to mean activities which result in emissions to air, land or water, or which have a potential health impact, especially where these provide a benefit to society as a whole. Examples include emissions from road traffic, sewage treatment and electrical power generation)
- iv. Where possible, assess the relative importance and risks of different impacts, drawing upon evidence of public risk perceptions and preferences.
- v. Provide a clear, authoritative and quantitative assessment of the impacts
- vi. Draw objective conclusions from the evidence on the relative environmental and health impacts of different waste disposal options, including an assessment of the robustness of these conclusions. These objective conclusions can then be used by Government to inform further waste policy decisions, including a consideration of the case for using economic instruments for incineration.

The scope of work was discussed further with DEFRA officials as the study progressed. It was agreed that, to allow the local effects of specific waste management facilities to be characterised, the report would focus on the direct health and environmental effects of the individual facilities. The report would not address life-cycle issues to the same level of detail.

It was agreed that the report would consider the currently operating or available options for waste management. The report does not cover industrial reprocessing facilities for materials removed from the MSW stream; however, it does cover materials recycling facilities where MSW is sorted for recycling and disposal.

It was agreed that the report could not study in detail emissions under abnormal operating conditions. This is firstly because of a lack of information on emissions under abnormal conditions. Secondly, in discharging their obligations with regard to the planning process, DEFRA and Local Authorities work on the basis that processes will be operated in compliance with their permits/licences. The licensing



Introduction	Review of on Emiss	of Information sions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qui Env Cor Em	antification of the vironmental nsequences of nissions	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions
Background Spe		ecification	Sources and types of waste		Health eff	ects	Pro	ocesses of exposure	

authority needs to be satisfied that a process operator can comply with the terms of its permit/licence. However, field data on the health or environmental effects of waste management operations in general reflects the effects of operating under both normal and abnormal operating conditions.

22

Introduction	oduction H		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2. REVIEW OF INFORMATION ON EMISSIONS

23

Introduction	ntroduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Summary – information on emissions

This Chapter of the report reviews available information on the emissions to air, land and water, including the amounts of solid residues, from waste management facilities. Where possible, we have estimated the amounts of emissions – for example, in many cases, we have provided an estimated release rate of pollutants per tonne of waste processed.

Alongside the estimated amounts of pollutants emitted from waste management facilities, we have described the data quality as very good, good, moderate or poor.

Emissions to air occur from most if not all waste management processes. Waste management facilities are designed, assessed and monitored to ensure that any impacts of emissions to air are at an acceptable level. Emissions to water occur in particular from landfill and composting, mostly via treatment at a sewage works. The main routes for disposal of MSW solid residues (other than to landfill) is spreading of composted material to land, and recycling of incinerator ash and similar materials from related processes.

The estimated emissions data is summarised in Section 2.14. This section draws together the information gathered for each emission from each type of facility. We recommend that readers refer to the main body of the report to understand the source and limitations of individual data values.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.1 Introduction

This Chapter of the report reviews available information on the emissions to air, land and water from the agreed range of waste management facilities. Emphasis is placed on data from UK operations where possible.

Where possible, we have quantified emissions from each waste management option to relevant media. Where sufficient data are available a release rate for a pollutant per tonne of MSW processed at the waste treatment/disposal has been derived. This information along with the methodology and assumptions used during the assessment are presented below.

The data underlying the assessment of emissions from waste management operations are less than ideal in many respects. In many cases, the available information is limited in coverage, of uncertain quality, and does not specify the details of the process and waste to which it refers. This sets a limit on the confidence that can be placed in the data. This is reflected in the assessment of uncertainty inherent in the emissions estimates in this chapter. We provide both an estimated uncertainty range, and also an evaluation of the "pedigree" of the data – that is, the reliability of the information underlying the emissions estimates. We also make recommendations for areas where future work would be most valuable to address the current shortcomings in data availability.

Material flows within the waste management framework are illustrated in Figure 1.5. Where possible, we have provided estimates of pollutant emissions for the key release points to the environment illustrated in Figure 1.5.

Emissions to air occur from most if not all waste management processes. Following treatment and discharge, emissions to air are no longer subject to control. Therefore facilities are designed, assessed and monitored to ensure that any impacts of emissions to air are at an acceptable level.

Emissions to water occur in particular from landfill and composting, via discharge to the sewerage system often after on-site treatment, prior to treatment in off-site sewage treatment works. Landfill leachate may also be released very slowly via the landfill lining system. At a limited number of landfill sites, treated leachate is discharged to surface water. In contrast to airborne exposure pathways, levels of contaminants in water and food are controlled and regulated, reducing the likelihood of adverse health effects arising from these pathways.

The practices of spreading composted material to land, and recycling residues from incinerators, or pyrolysis/gasification processes, may open a pathway for leached contaminants to reach human or environmental receptors. The practices do not of themselves complete such a pathway. As noted above, controls on water and food quality limit reduce the likelihood of this occurring in practice.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.2 Scope of the Report

The report has collated evidence on emissions to air, land and water from the following waste management operations:

- Materials Recovery Facilities
- Composting (in-vessel)
- Composting (windrow)
- Mechanical biological treatment (MBT)
- Anaerobic digestion with energy recovery
- Gasification/pyrolysis with energy recovery
- Unsegregated incineration with energy recovery
- Incineration of pre-sorted wastes with energy recovery, typically at small scale
- Landfill with landfill gas flaring and/or energy recovery
- Waste transportation excluding collection

Although there are hundreds of substances emitted during waste management operations, most of these are released in very small amounts which, as far as is known, are harmless. The study focused on substances of concern, and those which are released in large quantities from the management of municipal solid waste. The pollutants assessed differ according to the receiving media, and are detailed below.

Emissions to air

The study focuses principally on the substances with limits on emissions to air specified in European Commission Directive 2000/76/EC, "The incineration of waste." Hence, the following substances are assessed, together with carbon dioxide (the number in brackets is the unique chemical Chemcial Abstracts Service [CAS] number):

Oxides of nitrogen	[Nitrogen dioxide 10102-44-0; nitric oxide 10102-43-9]
Particulates	[No CAS number]
Sulphur dioxide	[7446-09-5]
Hydrogen chloride	[7647-01-0]
Hydrogen fluoride	[73602-61-6]
Volatile organic compounds	[Substance group: no CAS number]
Dioxins and furans	[2,3,7,8-tetrachlorodibenzo-para-dioxin: 1746-01-6]
Metals: Cadmium	[7440-43-9]

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Nickel	[7440-02-0]
Arsenic	[7440-38-2]
Mercury	[7439-97-6]
Carbon dioxide	[124-38-9]

-

Additionally, substances associated with landfill emissions are considered, namely methane and five individual volatile organic compounds (VOCs). These substances are the most significant VOCs with regard to health impacts identified in a recent Environment Agency report investigating the trace components of landfill gas (*Environment Agency, 2002e*: Investigation of the composition and emissions of trace components in landfill gas).

Methane	[74-82-8]
1,1-Dichloroethane	[75-34-3]
Chloroethane	[75-00-3]
Chloroethene	[75-01-4]
Chlorobenzene	[108-90-7]
Tetrachlorethene	[127-18-4]

The greatest volume of data was found in relation to emissions to air. This is reflected in the data presented in this chapter. This emphasis on emissions to air in itself does not imply that the potential for health or environmental effects is necessarily greatest for air pollutants, but does highlight a possible need for consideration of emissions to or via other media in future research in this area. This part of the study drew in particular on information assembled as part of a research project carried out by Enviros Consulting Ltd for the National Society for Clean Air and Environmental Protection (NSCA, 2002).

An evaluation of electricity generated from MSW energy recovery processes is provided in Appendix 3. This may be helpful in evaluating the significance of greenhouse gas emissions in wider terms.

2.2.1 Emissions to land/groundwater/surface water

The evaluation of emissions from landfill considers those substances found in more than 5% of samples of landfill leachate in Environment Agency research (*Environment Agency, 2003h*), "Updating the landfill leachate pollution inventory tool." Enviros Consulting Ltd, Shrewsbury). These comprise:

Aniline	[62-53-3]
Methyl tertiary butyl ether	[1634-04-4]
Chloride	[16887-00-6]
Cyanide	[57-12-5]

Introduction Scope Sources Recovery Composting Composting (underseal) Composting (underseal) Composting (underseal) Composting (underseal) Composting Comp	Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Facilities (Infreduct) (Window) treatment digeston flyotysis and incirculation floor	Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

Di(2ethyl hexyl)phthalate	[117-81-7]
Ethylbenzene	[100-41-4]
Fluoride	[16984-48-8]
Methyl chlorophenoxy acetic acid	[94-74-6]
Dichloromethane	[75-09-2]
Nitrate	[7697-37-2]
Organo-tin	[Substance group: no CAS number]
Phenol	[108-95-2]
Phosphorus	[7723-14-0]
Polycyclic aromatic hydrocarbons	[Benzo[a]pyrene: 50-32-8]

Nonyl phenol	[25154-52-3]
	[]

The evaluation of emissions to water from incineration comprises those substances specified in the Waste Incineration Directive (WID) – namely, nine metals, total suspended solids and dioxins and furans:

Dioxins and furans	[2,3,7,8-tetrachlorodibenzo-para-dioxin: 1746-01-6]
Cadmium	[7440-43-9]
Nickel	[7440-02-0]
Arsenic	[7440-38-2]
Mercury	[7439-97-6]
Thallium	[7440-28-0]
Lead	[7439-92-1]
Chromium	[7440-47-3]
Copper	[7440-50-8]
Zinc	[7440-66-6]

All identified information on releases from other processes is reported.

2.2.2 Data pedigree

"Pedigree" is a description of the reliability of the information from which the numerical data have been derived (Funtowicz and Ravetz, 1990). It goes beyond the numerical evaluations described below, to consider the following aspects of the quality of information (derived from van der Sluijs et al., 2002):
Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

- Proxy is the value based on a direct measurement of the parameter in question, or on some other measurement which is correlated more or less well with the parameter?
- Empirical basis is the value based on a large number of field measurements, a smaller number of field measurements, modelled values, estimates or speculation?
- Methodological rigour is the data obtained using best practice, widely used approaches, laboratory or research tools, or is no information provided on these methods?
- Validation can the data be cross-checked extensively, to a limited or indirect extent, or not at all?

The Pedigree of the emissions data set out in Chapter 4 was estimated by scoring key elements of the underlying data between 0 and 4 on the above four aspects, using the framework described by van der Sluijs et al. (2002). The key inputs were identified, and the Pedigree was established from the lowest score for any of these key inputs. A score of 0 - 4 was described as "poor", 5 - 8 "moderate"; 9 - 12 "good" and 13 - 16 "very good".

References are generally listed with the section on each waste management facility, as well as in Appendix 1. References which are published in the peer reviewed literature are marked in **bold**. References which have been reviewed by other means (e.g. Environment Agency research reports, or information published by reputable governmental bodies) are marked in *italics*. Other references are marked in normal type.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.3 Sources of Information

The main sources of information used within this Chapter are detailed below. Each section details specific references used, whilst a complete list of all sources reviewed during the compilation of the Chapter is provided in the Bibliography.

- Industry returns to the Pollution Inventory for 2000, 2001 and 2002 (where available).
- Environment Agency/DEFRA sponsored research (e.g. research into greenhouse gas emissions from UK landfills)
- Operator data
- Published research
- NSCA "Comparison of Emissions from Waste Management Options" report

Estimates developed specifically for this project (e.g. rate of generation and seepage of landfill leachate from different types of landfill).

Introduction		Review on Emi	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.4 Materials Recovery Facilities

2.4.1 Introduction

There are a variety of types and purposes of Materials Recovery Facilities (also known as Materials Recycling Facilities or MRFs) which may range from bulking up of source separated fractions of the waste stream with limited sorting and processing taking place, to more advanced mechanical or manual separation processes of mixed recyclables. MRFs may be attached to, or incorporated within, transfer stations or other waste facilities or may be separate dedicated facilities dealing purely with the recyclable fraction of collected municipal waste. Some Materials Recovery Facilities incorporate a degree of materials processing to enhance the quality or value of the materials extracted from the municipal stream (for example, by granulating plastics on site). Materials Recovery Facilities in the UK do not handle green or putrescible elements of household waste and are considered to process 'dry' recyclables only (e.g. paper, plastics, glass, metals, textiles etc) these are also known as 'clean' MRFs. Due to market acceptability of recyclate and operational experience over the last ten years in the UK, it is unlikely that Materials Recovery Facilities processing organic wastes or 'dirty' MRFs will find any significant future application in the UK and this study does not address the health or emissions impacts of this type of facility.

2.4.2 Sources of data; limitations; assumptions; areas of uncertainty

Emissions from the plant are likely to be primarily fugitive emissions to the air from waste handling and sorting. The only residue to land is likely to be the reject fraction from the process which would go to landfill or incineration. There are no leachate and limited water emission impacts from clean MRFs unless further processing of materials takes place on the same site.

The key aspect of controlling emissions from MRFs relates to the management of incoming wastes to ensure that wastes are suitable for handing at a MRF, and that the residence time of wastes in a MRF is minimised. Assuming the materials are representative of MSW, then abnormal emissions could potentially arise if waste remains in the facility for an extended period of time. This could give rise to increased emissions of odour, and possibly micro-organisms.

2.4.3 Emissions to air per tonne of waste

By their nature, Materials Recovery Facilities provide an opportunity for materials in the waste stream to be recycled, reducing the need to use raw materials. Reprocessing materials in this way could result in increases or decreases in emissions and effects at locations remote from the MRF itself. Reprocessing recycled materials often has associated environmental emissions – for example, if more heat is needed for reprocessing compared to that which would otherwise be required, or if long transportation distances are required. In other cases (such as the use of recycled glass cullet), recycling can require less energy or result in lower emissions than is required for manufacture using new materials. Also, if recycling results in a reduction in demand for raw materials, this would result in lower emissions associated with the extraction of raw materials. These complex issues lie outside the scope of this report, which focuses on the potential emissions and effects associated with the facility itself.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Emissions from Materials Recovery Facilities are fugitive in nature and arise during the tipping and sorting process. The processing of recyclables in an MRF occurs in a housed facility which may operate under negative pressure to allow the cleaning of the air emitted from the facility through the use of a bio-filter or other control mechanism.

The record of quantifiable emissions data for Materials Recovery Facilities is limited. A US Environmental Protection Agency (EPA) study into Environmental, Economic and Energy impacts of MRFs (MRF 1) addressed air emissions from case studies in the United States. The study considered total suspended particulates, fine particles (known as PM₁₀), lead, carbon monoxide, mercury vapour and volatile organic compounds (VOCs). For occupational and public health reasons bioaerosols, dust, silica, metals, polychlorinated biphenyls (PCBs) and pesticides were also assessed. Whilst there have been observed occupational health issues associated with respiration and manual handling of mixed recyclables (MRF2, MRF4, see Chapters 5 & 6) in terms of environmental emissions impact, studies have shown varied and overall poorly quantified impacts (MRF 1).

In the US EPA study a series of monitoring points were set around various sites to establish measurements of pollutants arising from the MRF operations. As noted above the data were not substantive and may have been influenced by other factors such as vehicle movements and the proximity of other waste related facilities nearby. The overall environmental impact of waste facilities is included in Chapter 6.

2.4.4 Emissions to land/groundwater per tonne of waste

The only direct emission from Materials Recovery Facilities to land is through the landfilling of the reject fraction of the plant. This fraction will vary depending on the type of collection systems in place which feed into the facility, the numbers of different materials being collected, the effectiveness of separation by the householder and the efficiency and configuration of the plant operation itself. Typically, reject fractions are in the order of 5 - 15% of the input material (however due the variables noted above may be considerably at variance with this range) and comprise fines and contraries (e.g. bottle tops, contaminated recyclables, or products where no market may be available for a particular facility, e.g. carrier bags, yoghurt pots). Materials Recovery Facilities may handle a variety of sources of recyclables from co-mingled kerbside collected materials, to source segregated, bring derived or commercial and industrial wastes. The nature and proportion of these different sources together with the operation of the plant will determine the composition and the quantity of any residual fines at the end of the process (per tonne of input material).

2.4.5 Emissions to sewer/surface water per tonne of waste

The only water emission from MRFs is due to runoff from hardstanding and through any washing of vehicles and equipment, with the potential for minor spillage from crushing / baling of cans / bottles. The US EPA study (MRF 1) showed the levels to be well below any environmental control limits and this aspect is not a significant issue for 'clean' Materials Recovery Facilities.

2.4.6 UK Emissions from materials recovery facilities

There are around 90 Materials Recovery Facilities operating in the UK at present handling municipal wastes. A national estimate of emissions cannot be made, because of the absence of quantifiable environmental emissions.

Introduction		Review on Emi	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & ic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.4.7 Data sources

Reference No	Reference Document
MRF 1	EPA, 1995. Environmental, Economic and Energy Impacts of Materials Recovery Facilities. National Risk Management Research Laboratory.
MRF 2	Gladding T. & Crook B., 2000 Development of Dust Monitoring Guidelines for Licensed Waste Management Facilities R&D Technical ReportP399
MRF 3	Wheeler P.A. and Rome L. de, 2002. Waste Pre-Treatment: A Review R&D Technical Report P1-344
MRF 4	Gladding T.L. et al. Air Quality and Worker Symptoms in Nine Materials Recovery Facilities (MRFs) in England and Wales Integrated Waste Management and Pollution Control: Research, Policy and Practice
MRF 5	Environment Agency, 2003g Monitoring of Particulate Matter in Ambient Air around Waste Facilities Technical Guidance Document M17
MRF 6	Institute of Wastes Management, 2000. Materials Recovery Facilities, IWM Business Services Ltd.
MRF 7	Materials Recycling Week, 2003. Materials Recycling Handbook. EMAP Communications.
MRF 8	Environment Agency, 2002g. Waste Pre-Treatment: A Review Report P1-344.

33

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.5 Composting (In-vessel)

2.5.1 Introduction

In vessel composting encompasses a wide range of techniques for the composting of organic materials in encapsulated environments. It is suitable for a wider range of organic waste materials, than 'open' composting including food processing and catering waste. This is due to the enclosed nature of the process which can be controlled and monitored to develop a high enough temperature throughout the vessel for a sufficient amount of time to ensure the required level of pathogen kill. A risk assessment of composting to achieve conditions for pathogen destruction has been undertaken on behalf of DEFRA (IVC 1), and under the Animal By-products Regulation 2003 and the EU Animal By Products Regulation (Regulation (EC) 1774/2002). The requirement that any 'catering type' wastes, including food/kitchen wastes from domestic premises are processed in in-vessel or in enclosed composting systems has been established.

It is a developing technology and trials are currently underway to assess its suitability for municipal wastes, in particular, source-segregated organic wastes. An in-vessel system at Alfreton has been built to process mixed waste. In addition, Island Waste operates an in-vessel system at Lynnbottom on the Isle of Wight which processes green waste, source-separated household organics and has processed refuse-derived fuel production fines. An in-vessel system at Ipswich processes green waste and sewage sludge. At this stage it appears likely that the majority of new In-vessel systems will be of the batch processing variety utilising tunnels for composting. New requirements in relation to animal by-products have increased the impetus for in-vessel composting systems as an aid to meeting the requirements of the biodegradable landfill diversion targets of the EC Directive on the Landfilling of Waste.

The use of in vessel composting is therefore likely to increase. Local Authorities, under pressure from waste recycling and composting targets and landfill diversion obligations are looking toward composting of source-segregated organics to help meet targets set out in the national waste strategies.

Gaseous emissions from in-vessel systems consist predominantly of carbon dioxide, water vapour and potentially small quantities of ammonia and some volatile organic compounds and bioaerosols (fungi, bacteria, actinomycetes, endotoxin, mycotoxins, and glucans) (IVC 2). Particulates will also be released. Inhalation of organic dust can cause a range of immunological respiratory symptoms. Of the substances considered in this study, composting systems will also give rise to emissions of carbon dioxide and methane (it is impossible to guarantee that all material will be kept under aerobic conditions at all times).

2.5.2 Sources of data; limitations; assumptions; areas of uncertainty

Any emissions from composting processes will vary according to the types of waste input and the operational procedures in place at the composting facility (e.g. addition of water to the process, nature of shredding process, residence time etc). Quantitative data on emissions from in vessel composting facilities is sparse due to the relatively new expansion into this area for municipal derived organic waste processing. Therefore the existing data does not allow figures for comparison with other disposal or treatment facilities to be derived. The majority of work focusing on emissions from composting sites, including those described below and others reviewed, concentrate on measurements of ambient concentrations. There is a lack

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

of accurate measurements of emissions, even from composting systems where the source is reasonably well defined such as an in-vessel system. No quantitative estimates are therefore provided for emission to air from composting systems, therefore a qualitative study has been developed of experience in this area.

The Environment Agency (IVC 3) commissioned a study to investigate emissions and associated health effects at three compost sites, one of which was an in-vessel system processing mixed green waste, source separated household organic waste and refuse derived fuel production fines. This study informs part of this discussion.

The impacts to land from the composting process arise from the application of the compost. A qualitative study of the data for mixed green and putrescible household waste streams has been used for this analysis. Due to the lack of data on In-vessel systems processing municipal derived organic wastes the analysis is based upon data from windrow systems processing these waste streams which will be similar in composition of the residue from in-vessel systems and the quality of the composting process is primarily dependent on the quality of the feedstock (IVC 7). The kitchen (food) waste element tends to have a higher moisture content than green wastes at about 80% rather than the 50% moisture content of green wastes. This will clearly impact on the relative mass reductions of processing the two waste streams. For the purpose of this exercise a mass reduction of 40% has been assumed for a green waste only feedstock (through windrow composting – IVC 8) and a reduction of 50% assumed for mixed municipal organic waste feedstock, through in-vessel systems. The amounts of kitchen waste entering the system will be partly dependent on the effectiveness of the collection system in place.

The emissions to water arise from the generation of leachate during the composting process. Data included in this section is an analysis of the liquor produced from a UK plant, however it may not be wholly indicative of a typical composition due to the processing of Refuse Derived Fuel fines as part of the feedstock. Most in-vessel systems re-circulate leachate within the process and have a net liquid loss throughout the system. Discharges to water will only arise from those systems with a net liquid excess (IVC 10).

Emissions under non-standard conditions are discussed together with windrow composting in Section 2.6 below.

2.5.3 UK Emissions from in-vessel composting

The 1999 Composting Association Survey (2001) (IVC 12) recorded a total of 32,717 tonnes of material composted by In-Vessel systems in 1999. This data informs the data in Table 2.1 and assumes that this tonnage is wholly comprised of organic MSW. The quantity of kitchen and garden waste passing through Invessel type composting systems is likely to have increased since the developments with Animal By-products legislation and this trend is likely to continue as landfill diversion and recycling targets become more stringent.

2.5.4 Emissions to air

Emissions to air from in-vessel systems will be affected by any systems for abatement and control of emissions. These may control emissions of microorganisms (e.g. via a liquid spray) and/or volatile organic compounds (e.g. via a carbon filter or bio-filtration system).

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Levels of bacteria recorded in the vicinity of the in-vessel system monitored in the EA study exceeded 10^7 colony forming units per cubic metre of air (cfu/m³) air¹. As is to be expected, unloading of the vessel gave rise to the highest levels of airborne micro-organisms recorded at the site. It should be noted that a landfill site and civic amenity site are located close to the in-vessel system which may have influenced the recorded levels. The most significant species of concern regarding waste management facilities are gram negative bacteria, aspergillus and penicillium (Crook and Swan, 2001).

To place these measurements in context, Swan et al (IVC 2) reviewed ambient levels in the absence of any significant bioaerosol sources and concluded that ambient levels are about 10 to 10⁶ times lower than those recorded during the handling of compost. There is clearly a wide range of variation in this data, this is partly attributable to operational environments and different procedures. The Environment Agency (2001a) concluded that appropriate conservative ambient levels to aim for to reduce the possibility of health effects are between 300-1000 cfu/m³. Concentrations of inhalable dust recorded by the Environment Agency (IVC 3) at the in-vessel system ranged up to 9900µg/m³. Volatile organic carbons (VOCs) were also measured at the in-vessel system site. Only 5 compounds were detected (toluene, ethylbenzene, mp-xylene, o-xylene, and 2-butanone) although the study authors note that the measurements may include a contribution from traffic. A study to characterise VOC emissions from a range of types of MSW composting facilities in the USA identified approximately 35 species (IVC 4). No information is provided regarding those specifically identified in the vicinity of invessel systems.

Work carried out as part of the development of the life cycle tool, WISARD, (IVC 5) has also been reviewed. Emissions data are presented for some composting processes however the data provided are not for in-vessel systems and have therefore not been included in this element of the study (IVC 6).

For emissions to land and water, there is some data composition of composts produced and leachate from the composting process which, whilst limited to a relatively small dataset, is included in the following sections.

2.5.5 Emissions to land/groundwater per tonne of waste

Emissions to land from composting process arise from the chemical content of the composts produced. The composition of the compost will be dependent upon the materials processed, the operational aspects of the plant, residence time, temperature etc. The nature of compost from in-vessel and windrow systems are comparable depending upon the feedstock (IVC 7). The developments with Animal By-Products legislation as noted previously prevents kitchen and other food wastes from being utilised in windrow systems, therefore an incentive of in-vessel systems is that they can tackle feedstocks containing both 'green' and 'kitchen' elements of municipal wastes. An overall 50% overall mass reduction has been assumed for in-vessel processing (see section 2.6.2).

Table 2.1 shows the compositions of composts produced through processing the organic component of household wastes with varying proportions of garden to kitchen wastes. These were processed in windrow type compost plant; however, the composition is likely to be similar to that produced through in-vessel systems where there is a limited dataset.



¹ cfu – colony forming units – a unit of measure for micro-organism numbers based on the growth of bacteria to form colonies on nutrient plates that can be subsequently counted.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequence Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Emissions to	Emissio processed (ns per tonne of g/T except whe	f waste ere stated)	UK emissions (kg / year	Data Pedigree
Land	Minimum	'Best Estimate'	Maximum	except where stated)	
Arsenic (As)	1.5	1.75	2	57	Moderate (8)
Cadmium (Cd)	0.15	0.7	1.15	21	Moderate (8)
Mercury (Hg)	0.005	0.25	0.5	8	Moderate (8)
Nickel (Ni)	1.5	6.8	33.5	220	Moderate (8)
Lead (Pb)	8.5	70	130	2300	Moderate (8)
Chromium (Cr)	3.25	28	167	910	Moderate (8)
Copper (Cu)	6	53	144	1700	Moderate (8)
Zinc (Zn)	39	117	365	3800	Moderate (8)
Magnesium (Mg) [7439-95-4]	390	420	450	14000	Moderate (7)
Ammonium [14789-03-9]	0	2.24	232	1300	Moderate (7)
Dioxins/ Furans (PCDD/F) ngTEQ/t	3.1 ng TEQ/T	7.2 ng TEQ/T	11.3 ng TEQ/T	0.23 mgTEQ/year	Moderate (7)

Assumption: 50% mass reduction through composting process T: tonnes

TEQ: Toxic equivalent

It should be noted that this dataset may not be representative and is only a subset of substances of concern.

2.5.6 Emissions to sewer/surface water per tonne of waste

Composting processes will produce some liquid residue during decomposition. There is a wide variation in levels of liquid recorded and it is important, with external processes, to differentiate between leachate liquid produced by the process and rainfall runoff from the site itself. The data in Table 2.2 shows quantities of liquid leachate derived from different composting processes (IVC 11), this table has been derived from a single reference source. Some in-vessel systems re-circulate liquid leachate, whilst others treat the liquid residue if required or discharge it direct to sewer if appropriate. The Composting Association Guide on In-Vessel Composting (IVC 10) states that up the 30% of the feedstock weight may be released as water (primarily vapour) during composting, however many in-vessel systems have a negative water balance due to the heat produced during the composting and so may add water to the process.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.2	Typical levels of lea	achate production from	different composting systems

Composting system type	Leachate Production range (Litres/T feedstock)	Data Pedigree
Passively aerated piles*	14 - 60	Poor (4)
Passively aerated rotating drums	48-63	Poor (4)
Rotating drums featuring forced, negative pressure aeration	44-56	Poor (4)
Static in-vessel featuring forced aeration	100-200 (may be re- circulated)	Poor (4)

Source: IVC 11 * This is not normally considered to be a correct composting procedure as composting is a managed process.

An analysis of the liquid from the tray wash and vessel sump from in vessel processing of green waste, source separated household organics and refuse derived fuel (RDF) fines gave the data in Table 2.3 (IVC 3). This water is reused in the process, therefore in this case there would not be an impact on the water environment. The processing of RDF fines in this dataset also makes it unusual. It should be noted that the longer the duration of the composting process the lower the biological and chemical oxygen demand (BOD and COD) are likely to be.

Analyte	Units	Tray wash water Sample 1	Tray wash water Sample 2	Vessel sump liquor
Cadmium (Cd)	mg/L	<0.01	<0.01	0.06
Chromium (Cr)	mg/L	<0.6	<0.6	0.29
Copper (Cu)	mg/L	<0.009	<0.009	0.2
Mercury (Hg)	mg/L	<0.06	<0.06	0.34
Lead (Pb)	mg/L	<0.1	<0.1	0.44
Zinc (Zn)	mg/L	0.06	0.16	1.8
Cyanide	mg/L	0.01	<0.01	<0.01
Suspended solids	mg/L	180	30	312
Monohydric Phenols	mg/L	0.15	0.09	<1
Chloride	mg/L	55	40	9130
COD	mg/L	580	360	270,000
BOD	mg/L	70	20	108,000
рН		7.3	7.7	6.3

 Table 2.3
 Analysis of liquid leachate from an in-vessel composting system

L: Litre

Assuming that 150 litres of liquid leachate is produced per tonne of input material through the in vessel composting process then the mass of substance released per tonne input can be calculated. The upper limit of the uncertainty range is shown in Table 2.4. In practice, some or all of the leachate may be re-circulated in the process. Consequently, the minimum value in Table 2.4 is set to zero, and the best estimate at half the upper limit value.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.4	Quantity of	pollutants in	leachates from	In-vessel	composting
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Emissions to Water	Emissions per	tonne of waste pro	cessed (mg/T)	Data
works	Minimum	'Best Estimate'	Maximum	Pedigree
Cadmium (Cd)	0	4.5	9	Poor (4)
Chromium (Cr)	0	23	45	Poor (4)
Copper (Cu)	0	15	30	Poor (4)
Mercury (Hg)	0	26	51	Poor (4)
Lead (Pb)	0	33	66	Poor (4)
Zinc (Zn)	0	135	270	Poor (4)
Cyanide	0	0.75	1.5	Poor (4)
Suspended solids	0	23400	46800	Poor (4)
Monohydric Phenols	0	11	23	Poor (4)
Chloride	0	685000	1370000	Poor (4)
рН	6.3	6.8	7.3	Poor (4)

The significant variation in compositions outlined in Table 2.4 may be attributed to the point where leachate is extracted from the process, in addition to the process factors of residence time, composition of feedstock etc. It should be noted that in many processes the liquid leachate is re-circulated and so a release into the water environment may not occur.

Given this level of variation and the relatively low number of systems on the UK market it has not been appropriate to extrapolate these impacts to form a national impact based on the limited dataset currently available. It should also be noted that this sample is based on a process using RDF fines as part of the feedstock.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.5.7 Data sources

Reference No	Reference Document
IVC 1	DEFRA, 2002. Risk Assessment: Use of Composting & Biogas Treatment to Dispose of Catering Waste containing Meat
IVC 2	Swan JRM, Crook B, Gilbert EJ, 2002. Microbial Emissions from Composting Sites, Issues in Environmental Science & Technology, Vol 18, Environmental Impact of Solid Waste Management Issues
IVC 3	Environment Agency, 2001a. Health Effects of Composting A Study of Three Compost Sites and Review of Past Data Technical Report P1-315/TR
IVC 4	Eitzer,1995. Emissions of Volatile Organic Chemicals from Municipal Waste Composting Facilities, Environmental Science and Technology, Vol. 29, No. 4, pp896-902
IVC 5	Environment Agency, 2000a. Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion, R&D Project Record P1/392/4
IVC 6	National Society for Clean Air & the Environment (NSCA), 2002. Comparison of Emissions from Waste Management Options
IVC 7	Information held by Enviros
IVC 8	Composting Association, Email from Dr Jane Gilbert to Enviros, 7th May 2003
IVC 9	National Assembly for Wales, 2003. Pilot Study on Municipal Waste Composition in Wales.
IVC 10	The Composting Association, Directory of In-Vessel Systems
IVC 11	CIWM, 2003. Biological Techniques in Solid Waste Management and Land Remediation
IVC 12	Slater R.A., Frederickson, J & Gilbert, E.J. 2001. The State of Composting1999 – The Results of the Composting Association's Survey of UK Composting Facilities & Collection Systems in 1999. The Composting Association.
IVC 13	Amlinger, F, Heavy Metal Thresholds in compost with a view to soil protection – what approach for the Community?, Report to the European Commission
IVC 14	Johansson C. et al. 1997 Compost Quality and Potential for Use, AFR Report, Heavy Metals in Compost Paper Swedish Environmental Protection Agency
IVC 15	The Composting Association as cited in CIWM Guide, 2003
IVC 16	Environment Agency, 1997a. A review of Dioxin Releases to Land and Water in the UK
IVC 17	Environment Agency, 2001d. Monitoring the Environmental Impacts of Waste Composting Plants. Technical Report P428.
IVC 18	Environment Agency, 2003g Monitoring of Particulate Matter in Ambient Air around Waste Facilities Technical Guidance Document M17
IVC 19	London Borough of Bexley, 2003. Bexley Kerbside Organics Collection and Processing Project, Summary report.
IVC 20	Henry Doubleday Research Association, 2001. Compost Analysis & Testing service (CATS)



Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.6 Composting (Windrow)

2.6.1 Introduction

Windrow composting comprises the aerobic processing of organic wastes placed in rows and either actively aerated or turned to promote aeration and decomposition of the material to form compost. It is a technique now confined to the green (or garden) waste element of MSW due to recent legislation on Animal By-Products (see previously). The process may be covered or take place in the open air. The process can also take place in static piles aerated through either a 'sucking' or 'blowing' action. This removes the need for turning in order to provide aeration.

Windrow composting is another waste management option increasingly being used as part of local waste strategies to help meet the obligations of the Landfill Directive and the targets of the National Waste Strategies.

2.6.2 Sources of data; limitations; assumptions; areas of uncertainty

As discussed in section 2.5.2 there is a lack of accurate measurements for emissions to air from composting processes. The following data is included for reference but as the emissions to air are usually fugitive in nature, quantifying release per tonne of input waste is prone to wider margins of error (WC 2). Additional data is included however on emissions to land and water, where the data is more comprehensive.

Emissions to land from composting processes arise from the composition of the composts produced. The composition of the compost will be dependent upon the materials processed, the operational aspects of the plant, residence time, temperature etc. The developments with Animal By-Products legislation (as noted previously) prevents kitchen and other food wastes from being utilised in windrow systems. The kitchen, food waste element tends to have a higher moisture content than green wastes at ~ 80% rather than the ~50% moisture content of green wastes. For the purpose of this exercise a mass reduction of 40% has been assumed for green waste feedstock.

For water impacts from composting processes, a median of 24 litres of leachate derived from a range of 14 - 34 litres anticipated to arise through the composting process (WC 3) has been used. This excludes the impact of runoff from the hard-standing areas which is dependent upon rainfall and the size and drainage of the site, and therefore represents the moisture released during decomposition of the organic matter under its own weight and that which is not released as water vapour.

2.6.3 UK Emissions from windrow composting

The Composting Association Survey of 1999 (2001) (WC 7) recorded a total of 618,517 tonnes of municipal sourced waste composted in 1999. As discussed windrow composting cannot now process kitchen waste and by excluding the collections including kitchen waste this will give a tonnage of 596,915 tonnes. This tonnage is used to inform Tables 2.5 and 2.6.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.6.4 Emissions to air per tonne of waste

Environmental emissions from windrow composting are normally fugitive in nature and therefore whilst point measurements can be taken the overall emissions are difficult to quantify accurately, particularly in relation to a tonnage of feedstock material. The Environment Agency as part of their work informing the Life Cycle Assessment tool WISARD (WC13), derived data on projected emissions to atmosphere of carbon dioxide, ammonia, particulates, chloride, water vapour and volatile organic compounds for composting processes which were related to input tonnage. Overall the current data is too inconsistent to inform an accurate 'gram per tonne' input emissions value. Some more generalised emissions data from another Environment Agency report (WC 5) also estimated an indicative CO_2 emission of around 10% of the intake tonnage from composting processes, or 100kg/T of input waste, however this would appear to be a low estimate. A report for the European Commission on the impacts of waste management facilities on climate change (WC15) produced a range of carbon dioxide emissions, related to time, carbon sequestration and relative to other waste management scenarios and a study (WC18) by White et al. suggested a CO_2 production of 323kg/T of feedstock from a windrow process. Clearly there is a significant level of variation amongst the existing literature in this area and illustrative outputs sourced from a variety of studies are included in Table 2.5. It should be noted that carbon dioxide emitted from composting processes is a net equivalent of zero production emissions as it represents part of the carbon cycle. Around 30% of volatile solids remain sequestered in the compost, which may be only slowly released to atmosphere over time.

Studies in the Netherlands of Ammonia emissions from Composting processes estimate that 240 grams of Ammonia per tonne of MSW (i.e. around 120 grams per tonne of input waste) are produced. The EC draft BREF Note on Waste Treatment (WC6) considered this estimate to be high for green waste only processes and more realistic for whole waste processes as a pre-treatment to landfill (see Mechanical Biological Treatment in this chapter). Other studies have (WC17) indicated that hydrogen sulphide may be emitted from composting processes.

Given the uncertainty over this area with regard to quantifiable environmental emissions from composting this section does not include a numerical summary table of emissions to air. However an indicative summary of measurements and emissions is included in Table 2.5.

Air Emission criteria	Indicative emission level / tonne of input material	Notes	Data Pedigree
Carbon Dioxide	100 – 482 kg/T	Higher figure based on per tonne of MSW (WC13)	Moderate (6)
Ammonia	5 – 120 g/T	Higher range only likely for whole waste composting	Poor (4)
Particulates	163 – 186 g/T	Bioaerosol monitoring data is included in chapter 5.	Poor (3)
Chloride	2 g/T		Poor (4)

 Table 2.5
 Indicative emissions to air from windrow composting sources

It is important to note that a key emission in terms of occupational and public health from Composting facilities is the release of bioaerosols, particularly during



Introduction		Review on Emis	Review of Information on Emissions		of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

turning/aerating of windrows. The most significant species of concern regarding waste management facilities are gram negative bacteria, aspergillus and penicillium.

Another aspect of increasing interest has been the study of VOCs from composting processes (Table 2.6). Further details on the health impacts of bioaerosols from composting processes are given in Chapter 3. Methane is likely to be produced in small quantities due to minor anaerobic activity, and carbon monoxide may also be released during the process.

Table 2.6	VOC emissions	from composting plant
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Compounds Detected	g/T MSW	Data Pedigree		
m,p Xylene [108-38-3; 106-42-3]	0.81	Moderate (6)		
Nonane [111-84-2]	0.44	Moderate (6)		
o Xylene [95-47-6]	0.54	Moderate (6)		
BetaPinene [127-91-3]	3.7	Moderate (6)		
Ocimene [13877-91-3]	3.0	Moderate (6)		
D-Limonene [5989-27-5]	10.5	Moderate (6)		
Undecane [1120-21-4]	2.4	Moderate (6)		
Dodecane [112-40-3]	1.2	Moderate (6)		
Methyl-(methylethyl)-Cyclohexane [99-82-1]	1.5	Moderate (6)		
Total	24.0	Moderate (6)		

Information from Ref. WC13

2.6.5 Emissions to land/groundwater per tonne of waste

The data in Table 2.7 is an assessment of the composition of compost following processing in a windrow type facility. The respective levels of pollutants and contraries and the physical properties of the compost will determine applicability to different end uses. The end use will determine potential pollution pathways and enable an assessment to be made of environmental impact. For example good quality composts, such as those that comply with the Publicly Available Specification (BS PAS 100) for compost, may have an agricultural or horticultural application whilst lower quality grades of compost may be used in some land restoration projects, or have application for daily cover on landfill sites. The latter will be an element in the pollution load of a landfill site, whilst the former will be applied directly to land. Other factors which can alter the composition of the compost are whether the source material is derived from an urban or rural environment (this can affect levels of polycyclic aromatic hydrocarbons and levels of dioxins and furans).

A study for the European Commission (WC15) noted that the reject level of contraries and contamination in municipally derived composts was in the region of 6% which may be sent directly to landfill.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.7	Heavy metals and pollutants in compost	derived from green wastes

Emissions to Land	Emissio p	ns per tonne rocessed (g/1	of waste ſ)	UK impacts	Data Refs	Data Pedigree
	Minimum	'Best Estimate'	Maximum	in kg/year		
Arsenic (As)	1.8	3.0	4.2	1,800	WC 9, WC 1	Moderate (8)
Cadmium (Cd)	0.22	0.53	0.84	320	WC 9, WC 1, WC 10	Moderate (8)
Mercury (Hg)	0.07	0.19	0.31	110	WC 9, WC 10	Moderate (8)
Nickel (Ni)	5.5	9.3	13.2	5,600	WC 9, WC 1, WC 10	Moderate (8)
Lead (Pb)	20.	67	114	40,000	WC 9, WC 1, WC 10	Moderate (8)
Chromium (Cr)	10	19	28	11,300	WC 1, WC 10	Moderate (8)
Copper (Cu)	17	29	41	17,400	WC 1, WC 10	Moderate (8)
Zinc (Zn)	80	130	174	76,000	WC 1, WC 10	Moderate (8)
Polychlorinated Biphenyls (PCB)	0.018	0.06	0.102	36	WC 11	Moderate (7)
Ammonium	0.0	2.2	230	72	WC 18	Moderate (7)
Dioxins/ Furans (PCDD/F) ngTEQ	1.2	17	34	18mg TEQ	WC 11	Moderate (7)

Based on assumption of 40% mass reduction during the composting process (Composting Association, 2003)

2.6.6 Emissions to sewer/surface water per tonne of waste

Table 2.8 below includes characteristics of a typical liquid leachate from systems composting source segregated household biodegradable waste (WC 3). Studies have shown (WC16) that leachate releases are most likely during the first 2 weeks of composting.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.8	Composition	of leachate	per tonne o	f waste processed
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Emissions to	Emissio processed	ns per tonne d in g/T, whe	of waste re relevant	UK impact	Data
Treatment works	Minimum	'Best Estimate'	Maximum	year	Pedigree
Ammonium	9.6	18	26.4	10744.5	Poor (4)
Total nitrogen	12.1	19.25	26.4	11490.6	Poor (4)
рН	5.7	8	10.3	-	Poor (4)
BOD5	240	672	1104	-	Poor (4)
COD	432	1032	1632	-	Poor (4)

Assumptions: 24 Litres leachate produced 14 - 34 litres / tonne feedstock (WC 8). Other information from Ref. WC3

2.6.7 Operation under non-standard conditions

The key issue which could potentially affect emissions from composting operations is the need to ensure aerobic conditions throughout the mix. In-vessel systems are normally designed to facilitate this process semi-automatically, whereas windrows need to be carefully managed and turned. This means that in-vessel systems are less likely to give rise to emissions under non standard conditions.

However, if good mixing does not take place, then increased releases of potentially odorous or hazardous volatile organic compounds and micro-organisms could occur from windrow or in-vessel systems, until aerobic conditions are restored. If air displaced from an in-vessel system is passed through an abatement system such as a biofilter, a baghouse filter or by passing the air through a co-located combustion facility, then these emissions can be controlled to avoid any significant increase in emissions even under these non-standard conditions.

Evidence on the health effects of composting facilities discussed in Chapter 3 suggests that such events could result in detectable acute symptoms from time to time in people living very close to MSW composting facilities. MSW composting processes are regulated by the Environment Agency, with the aim of minimising the occurrence of such incidents.

Reference No	Reference Document
WC 1	Environment Agency, 2001a. Health Effects of Composting A Study of Three Compost Sites and Review of Past Data Technical Report P1-315/TR
WC 2	National Society for Clean Air & the Environment (NSCA), 2002. Comparison of Emissions from Waste Management Options
WC 3	Chartered Institution of Wastes Management (CIWM), 2003. Biological Techniques in Solid Waste Management and Land Remediation
WC 4	Composting Association, cited in CIWM document WC 3
WC 5	Babtie Group Ltd, 2002. Environment Agency Pollution Inventory. Guidance on reporting emissions from waste management operations. Environment Agency.
WC 6	European Commission, 2003. Draft Reference Document on Best Available Techniques for the Waste Treatment Industries.

2.6.8 References

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequence Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Reference No	Reference Document
WC 7	Slater R.A., Frederickson, J & Gilbert, E.J. 2001. The State of Composting1999 – The Results of the Composting Association's Survey of UK Composting Facilities & Collection Systems in 1999. The Composting Association.
WC 8	Roth, T. 1991. Sickerwasser aus der Bioabfallkompostierung – Möglichkeiten der Behandlungund Ensorgung in einem dezentralen Anlagensystem. Dissertation, GH Kassel, Witzenhausen.
WC 9	Paulsen O, 1996 Genbrugsterminalen, Vejle commune, Denmark [Centre for Waste & Recycling, Municipality of Vejle]
WC 10	Amlinger, F, Heavy Metal Thresholds in compost with a view to soil protection – what approach for the Community?, Report to the European Commission
WC 11	Swedish EPA, 1997. Compost Quality and Potential for Use, AFR Report, Christina Johansson et al Volatile Organics in Compost Paper
WC 12	Swedish EPA, 1997. Compost Quality and Potential for Use, AFR Report, Christina Johansson et al Heavy Metals in Compost Paper
WC 13	Environment Agency, 2000a. Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion, R&D Project Record P1/392/4
WC 14	Environment Agency, 2003g. Monitoring of Particulate Matter in Ambient Air around Waste Facilities Technical Guidance Document M17
WC 15	Smith et al., 2001. Waste Management Options & Climate Change, report for the European Commission
WC 16	Enviros/ODPM (unpublished) Planning Issues Associated with Waste Management Facilities
WC 17	Tsiliyannis C. (1999) Report: Comparison of environmental impacts from solid waste treatment and disposal facilities <i>Waste management</i> & research 17(3)
WC 18	White P. <i>et al.</i> (1995) Integrated Solid Waste Management: A Lifecycle Inventory, <i>Blackie Academic and Professional</i>
WC 19	Henry Doubleday Research Association, 2001. Compost Analysis & Testing service (CATS)

46

Introduction		Review of Information on Emissions		Review Epidemi Researc	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.7 Mechanical Biological Treatment (MBT)

2.7.1 Introduction

Mechanical Biological Treatment (or whole waste composting) is a generic term for an integration of several processes individually assessed elsewhere in this chapter. MBT systems are usually designed such that emissions are minimised through housing under negative pressure and the use of bio-filters. Due to the variation in systems it is not possible to provide indicative emissions, however the references have provided the information included within this section which provide an illustration of particular systems and a limited dataset.

The MBT processes are usually designed to stabilise the residual waste stream after initial recyclables and compostables are removed or prepare it as a fuel for a thermal treatment process. The process may involve separation of the non compostable element of the waste stream through a sorting mechanism, for example magnetic separation of residual ferrous metals and the use of a homogenisation drum (including water injection) with screening of the output material into a reject fraction (comprising some textiles, plastics and metals with minor organic contaminants) and the remaining component windrow composting will be a stabilised residue which may be subject to further screening or further sorting (e.g. through air classification) dependent on the ultimate application of the residue. This applications if the residue is processed to a sufficient quality or may be fed into another treatment process such as incineration, RDF or gasification.

2.7.2 Sources of data; limitations; assumptions; areas of uncertainty

Emissions from MBT systems include impacts to air through emission of bioaerosols, dust, VOCs (see below) and odours. These will be fugitive in nature and dependent upon the waste inputs, meteorological conditions and the operation and configuration of the plant. The quantification of emissions related to a tonne of waste input is only practicable for the MBT solid residues (see below), however substantive or comparable data is presently limited on this area and the data shown below is derived from the development of the Best Available Technique Reference Note on waste treatment (MBT 1).

2.7.3 UK Emissions from mechanical biological treatment

At present there are no full scale Mechanical Biological Treatment Systems yet in operation in the UK, however there are systems under development and this is likely to be an area of expansion in the short to medium term to assist in the achievement of landfill diversion targets of biodegradable municipal waste.

2.7.4 Emissions to air per tonne of waste

The fugitive emissions to air from MBT processes are likely to be similar to those from other waste processing facilities such as Materials Recovery Facilities or Windrow Composting plant. The data however on MBT emissions is limited and not substantive and so a qualitative description is provided, with tables included to illustrate the emissions data and environmental monitoring information currently available.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Aerobic decomposition of input material through the MBT process can result (depending on the configuration of the system) in carbon dioxide and water being lost to atmosphere in a similar manner to windrow composting. One process (MBT 2) cites that for every tonne of input waste 200 - 250kg of water is evaporated. A study for the European Commission on Climate Change stated that around 22kg CO₂ equivalent per tonne of waste was produced through MBT processing. This would primarily be in the form of CO₂ emitted (MBT11).

The data cited in Table 2.9 is taken from the EC draft BREF note on waste treatment (MBT 1).

Emissions to Air	Emissio	ons per tonne of M	SW (g/T)	Data
Emissions to Air	Minimum	'Best Estimate'	Maximum	Pedigree
Methane (CH4)	No data	411	No data	Moderate (5)
Carbon Dioxide (CO2)	No data	181,000	No data	Moderate (5)
Carbon Monoxide (CO)	No data	72.3	No data	Moderate (5)
Hydrocarbons	No data	36	No data	Moderate (5)
Hydrogen Chloride (HCI)	No data	1.2	No data	Moderate (5)
Hydrogen Fluoride (HF)	No data	0.4	No data	Moderate (5)
Ammonia (NH3)	No data	~120	No data	Moderate (5)
Nitrogen Oxides (NOx)	No data	72.3	No data	Moderate (5)
Particulates	No data	No data	No data	
Oxides of Sulphur (SOx)	No data	28	No data	Moderate (5)
Dioxins/ Furans (PCDD/F) ng TEQ	No data	40	No data	Moderate (5)

 Table 2.9
 Emissions to air from mechanical biological treatment processes

Bioaerosols are also likely to be associated with these facilities. The most significant species of concern regarding waste management facilities are gram negative bacteria, aspergillus and penicillium. A further source of emissions to air from MBT systems may arise from any combustion of residues in an RDF system.

2.7.5 Emissions to land/groundwater per tonne of waste

Land impacts will depend on the level of processing of the residue material and its ultimate application. MBT is an interim process and consequently will have impacts associated with the waste management technique applied to the residue streams from the process. Emissions to land from MBT processing are the reject fraction from the system, which is usually sent to landfill, and the use of the primary solid residues which form the bulk of the output of the process (see Table 2.10). A stabilised residue from MBT sent to landfill has about 90% less landfill gas potential than raw MSW deposited into landfill (MBT11). Some systems are designed as a pre-treatment process for Refuse Derived Fuel incineration, Pyrolysis and Gasification or Anaerobic Digestion.

Data from one system (MBT2) suggests that approximately 17% of the input material is reject fraction in terms of fines. An overall mass output is included in Table 2.10 and is based on emission per tonne of MSW input. The fate of components is included and the impact to land will depend on the disposal or recycling application of the residues generated.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.10	Examples of solid residue emissions from one MBT process	
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	Emissions	s per tonne of I	MSW (kg/T)	Data		
Solid Residues	Minimum	'Best Estimate'	Maximum	Pedigree	Fate	
Refuse Derived Fuel	257.2	376.1	495	Moderate (5)	RDF Incineration process	
Residue from fines treatment	14.9	92.45	170	Moderate (5)	Landfill	
Low quality aggregate (fines, primarily glass and miscellaneous non combustibles)	80	82.1	84.2	Moderate (5)	Reused as a building material (provided process is optimised)	
Ferrous Metals	No data	50	No data	Moderate (5)	Recycled	
Non Ferrous Metals	No data	5	No data	Moderate (5)	Recycled	

Compositional data on these residue fractions is limited, and even those from a single process exhibit wide variation (Table 2.10) this is partly attributable to the varied nature of municipal wastes and also the variety of configurations of plant.

2.7.6 Emissions to sewer/surface water per tonne of waste

The water impacts are in some processes are minimal, as whilst water is commonly used in the initial processing it is then partially evaporated in the composting process. Leachate from the composting / biological drying process is re-circulated in some systems (MBT3) resulting in no overall waste water emission. The data included in Table 2.11 is from a system which results in a net waste water discharge (MBT1) and the composition is based on a gram per tonne MSW input basis. The net waste water discharge from this process is 261 litres per tonne of MSW.

Emissions to	Emissions	per tonne of MSW appropriate)	/ (g/T, where	Data	
Treatment works	Minimum	'Best Estimate'	Maximum	Pedigree	
Ammonia (NH ₃)	No data	160	No data	Moderate (5)	
Nitrates	No data	10	No data	Moderate (5)	
Sulphates	No data	5	No data	Moderate (5)	
COD	No data	530	No data	Moderate (5)	

 Table 2.11
 Waste water emissions from one MBT system

This waste water may be treated to varying degrees depending upon environmental permit requirements and an example of the output from a waste water treatment process on the waste water composition outlined in Table 2.11 is included in Table 2.12.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.12	Emissions to water from MBT waste water	processing	(MBT 1)	۱
		processing		,

Emissions to water	Advanced Waste Water Treatment (g/T, where appropriate) MSW	Data Pedigree
Ammonia (NH ₃)	20	Moderate (5)
Nitrates	1	Moderate (5)
Sulphates	1	Moderate (5)
COD	1	Moderate (5)

2.7.7 Operation under non-standard conditions

MBT systems are diverse, and emissions under non-standard conditions will depend on the make-up of the particular system. Possible failure scenarios include a failure to ensure aerobic conditions throughout the waste mix (as for composting processes). If aerobic conditions are not maintained, then increased releases of potentially odorous or hazardous volatile organic compounds could occur, until aerobic conditions are restored. A system which involves combustion of biogas could run the risk of emissions of unburnt biogas, in the event of a failure of the combustion system.

MBT processes would be regulated by the Environment Agency, with the aim of minimising the occurrence of such incidents.

Reference No	Reference Document
MBT 1	European Commission, 2003. Draft Reference Document on Best Available Techniques for the Waste Treatment Industries.
MBT 2	Industry Literature, EcoDeco
MBT 3	Enviros, 2002. Waste Technology Options – A review for Norfolk.
MBT 4	McLanaghan, S, 2002. Delivering the Landfill Directive: The role of new and emerging technologies.
MBT 5	Alternative Waste Management Technologies and Practices Inquiry, 2000. Report of the Alternative Waste Management Technologies and Practices Inquiry. The State Government of New South Wales.
MBT 6	Environment Agency, 2001a. Health Effects of Composting A Study of Three Compost Sites and Review of Past Data Technical Report P1- 315/TR
MBT 7	Environment Agency, 2002f. Waste Pre-Treatment: A Review Report P1-344.
MBT 8	Strategy Unit, 'Waste Not Want Not' report 2002. Annex G: Treatment & Disposal of Residual Waste – MBT in context.
MBT 9	Industry Literature, Herhof
MBT 10	Environment Agency, 2003g "Monitoring of Particulate Matter in Ambient Air around Waste Facilities" draft Technical Guidance Document M17
MBT 11	European Commission, 2001. Waste Management Options & Climate Change.

2.7.8 Data sources

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.8 Anaerobic Digestion with Energy Recovery

2.8.1 Introduction

Anaerobic Digestion is the process by which mixed microbiological cultures degrade organic material in the absence of oxygen resulting in the production of gas (principally methane and carbon dioxide) together with solid and liquid residues. Anaerobic Digestion is the process of degradation that occurs in landfill, however it can also be used as a managed process in enclosed vessels, where the feedstock is circulated, and usually heated using some of the gas produced. After this invessel processing the digestate is dewatered and the solid and liquid fractions treated as appropriate to their subsequent disposal or use. The solids for example can be composted aerobically to mature the residue for use on land.

Anaerobic Digestion (AD) is currently used in the UK to treat sewage sludge, some agricultural wastes and some industrial wastes and waste waters. It has been promoted as being suitable for the treatment of mixed organic municipal waste as well as source segregated organic waste in several European countries and there are successful commercial operations in several European countries using this technique for municipally derived organic wastes.

2.8.2 Sources of data; limitations; assumptions; areas of uncertainty

Although Thames Waste Management have trialled the anaerobic digestion of municipal organic waste with sewage sludge, there are currently no full scale anaerobic digestion plants in the UK (Biffa has, however, recently been awarded a contract with Leicester City Council to develop an AD solution). Consideration of emissions to air from anaerobic digesters therefore draws on a research study from Germany, where the technology is more widespread.

The information drawn here is based on a study undertaken during 1998 by the Oeko-Institute, Germany. The study compared the pollution potential and overall ecological burden of various waste technologies, in order to identify the least damaging option for a Waste Management Plan for the City of Műnster.

The anaerobic digestion plant set up for the German pilot study formed one stage in the treatment of waste:

- 1. Compostibles and clean recyclables are removed at source to leave a waste stream comparable to our classification of municipal solid waste residual fraction.
- 2. Biological and mechanical pre-treatment removes recyclables (metals, paper and plastic).
- 3. The remaining waste is input to the anaerobic digester.
- 4. The digester results in the production of biogas (fed to an engine-based energy utilisation plant) and heavy particulate sediment (sent to landfill).
- 5. The remaining digester residue is further treated via wet oxidation before being sent for landfilling or incineration.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

The anaerobic digestion was carried out in a one-step wet fermentation digester supplied by the Finnish company WABIO Technology. The first such system to be installed in Germany was in spring 1995 in Bottop.

Concentrations of substances in the biogas produced by the anaerobic digester were measured directly as part of the pilot project. In operational plants, biogas is not released directly to air during normal operation but is transferred to an energy utilisation plant. Key emissions to air from operating plants are those resulting from the utilisation plant. The study estimates concentrations that would result from the utilisation plant based on the following assumptions.

For combustion gases, emission concentrations resulting from the utilisation plant are assumed to be equal to those measured at a utilisation plant when burning landfill gas. For heavy metals (arsenic, cadmium, nickel and mercury) the measured concentrations from the utilisation plant are assumed to be equal to those in the biogas. For short-lived organics (e.g. tetrachloroethene), it was assumed that concentrations released from the utilisation plant will be a factor of 1000 less than those present in the biogas, (ie a destruction efficiency of 99.9% was assumed). In practice, destruction efficiencies will vary from plant to plant although a single value was assumed by the authors. It was assumed that the concentrations of persistent organic substances from the utilisation plant will be equal to those in the biogas. It was assumed in the Oeko Institute study that any substance where the concentration in the biogas was below the detection limit was not present in emissions from the utilisation plant.

In contrast to the approach used in the Oeko Institute study, for consistency with the rest of this assessment, where measurements or estimates of concentrations in emissions from the energy utilisation plant were at or below the limit of detection, we have assumed that the concentrations were equal to the limit of detection. This provides a worst case assessment.

2.8.3 UK emissions from anaerobic digestion facilities

There are no full scale Anaerobic Digestion facilities currently operating on MSW derived feedstock in the UK at present. Biffa Waste Services Ltd have a contract for developing an AD facility for Leicester City, but until such facilities are developed on a commercial scale for handling municipally derived wastes through Anaerobic Digestion the emission impact is negligible (only through smaller scale trials).

2.8.4 Emissions to air per tonne of waste

To provide anaerobic conditions, the vessel within which the process takes place is completely closed. Biogas is produced from the digestion process and this is usually used to heat the process and generate electricity, for example, by feeding the gas to an on-site combined heat and power (CHP) plant. Information is readily available on the composition of biogas (for example, Environment Agency, 2000f) however, biogas is not released direct to air under normal operating conditions. The key emissions to air are therefore those that result from the CHP plant.

Emissions data from the study are presented in Table 2.13. For each substance covered in this project for which measurements were made, the table shows:

- Measured emission concentrations in the biogas released from the anaerobic digester;
- Estimated emission concentrations released to air that would result from an associated energy utilisation plant;



Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

• Estimated mass release rates, per tonne of waste. These have been calculated from the post utilisation plant concentrations and the gas flow rate quoted in the study of 436 normalised cubic metres (Nm³)² per tonne of waste processed.

The study authors note that the measured emission concentrations in the biogas were subject to a large margin of error because the pilot project experienced varying gas input flows and temporary condensation which made representative sample extraction difficult. Additionally, due to the low gas volumes generated, multiple samples could not be taken concurrently. It is surprising to record concentrations of nitrogen oxides and sulphur dioxide in the biogas; these substances are closely associated with combustion processes and would not normally be expected to be present in biogas. The recorded levels may result from interference from other substances. No further information is supplied on the measurement technique used in the study.

Substance	Measured concentrations in Biogas (mg/Nm³)	Estimated concentrations from energy utilisation plant (mg/Nm ³)	Mass released per tonne of waste processed (g/T)	Data Pedigree
Nitrogen oxides	29	432 [†]	188	Moderate (8)
Particulates	No data	No data	No data	
Sulphur dioxide	44	6.9 [†] *	3.0	Moderate (8)
Hydrogen chloride	< 0.3	< 0.047 [†] *	< 0.02	Moderate (8)
Hydrogen fluoride	< 0.1	< 0.017 *	< 0.007	Moderate (8)
Total VOCs	No data	No data	No data	
1,1-Dichloroethane	No data	No data	No data	
Chloroethane	No data	No data	No data	
Chloroethene	< 2	0	0	Moderate (7)
Chlorobenzene	No data	No data	No data	
Tetrachlorethene	0.081	0.00081 [#]	0.0004	Moderate (7)
Methane	No data	No data	No data	
Cadmium	< 0.0018	< 2.81 x 10 ^{-4 §} *	< 0.0001	Moderate (8)
Nickel	< 0.0039	< 6.09 x 10 ^{-4 §} *	< 0.0003	Moderate (8)
Arsenic	< 0.0079	< 1.23 x 10 ^{-3 §} *	< 0.0005	Moderate (8)
Mercury	< 0.0083	< 1.30 x 10 ^{-3 §} *	< 0.0006	Moderate (8)
Dioxins and furans	< 0.0001 ngTEQ/Nm ³	No data	No data	
Dioxin-like PCBs	< 0.000042	No data	No data	
Carbon dioxide	No data	No data	No data	

 Table 2.13
 Emissions to air from anaerobic digestion plant

Source: AD 3 Note †:

†: Concentration measured in utilisation plant burning landfill gas.

§: Concentration assumed to be at the limit of detection.

#: Concentration assumed to be equal to 1% of the biogas value.

*: The estimated concentrations released from the utilising engine have been calculated using the same factor as for landfill gas utilising engines in Section 2.12.3



^{2 &}quot;Normalised" cubic metres refers to the volume that a gas sample would have if held at a specified set of reference conditions – for example, 0 degrees centigrade, 1 atmosphere pressure. In some cases, gas volumes are normalised to specific moisture level and oxygen content.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

As noted above, published data for anaerobic digestion processes are extremely limited. Derivation of emission estimates therefore link back to this single study. The uncertainties in the measurements are not quantified in the study and it is therefore difficult to quantify the uncertainty in using these values to represent emissions from any anaerobic digestion process. Where the estimate of emissions for a particular waste disposal option has been derived from a large number of operational plants, (as in the case of incineration) the variation in the measurements is considerable. It is therefore reasonable to assume that the variation in emissions across the range of anaerobic digestion systems would be similar.

The biogas is under positive pressure in the tank. It is therefore possible (albeit unlikely due to safety controls) that some fugitive emissions will arise. There is also commonly an aerobic maturation stage of the digestate which may release minor quantities of carbon dioxide and oxides of nitrogen as well as water vapour.

Offset to emissions from electricity generation

Anaerobic digestion would normally be operated with energy recovery, resulting in the generation of electricity for export to the National Grid. Generating electricity in this way may be viewed as reducing the need to generate electricity from other sources, with a benefit in reductions in emissions from these sources. The estimated reduction in emissions as a result of avoided use of fuels is set out in Appendix 3. This suggests that there would be a net reduction in emissions to air of sulphur dioxide, hydrogen chloride and metals from the generation of electricity by the use of anaerobic digestion, compared to emissions from UK power generation in 2001. Emissions of oxides of nitrogen from anaerobic digestion are similar to those from power generation in 2001.

2.8.5 Emissions to land/groundwater per tonne of waste

The emission to land from Anaerobic Digestion processes is the application of digestate to land as a soil conditioner, as landfill cover / restoration material or through land spreading (usually sewage sludge and agricultural based wastes). The quality and composition of the digestate depends on the input material and the type of Anaerobic Digestion system employed including any post processing maturation (e.g. composting of the digestate). Table 2.14 shows typical heavy metal composition from fractions of MSW. It should be noted that the land impacts are likely to depend on the input material and the level of processing. For example mixed MSW input is likely to go as a stabilised residue to landfill or another treatment / disposal option whereas source separated household organic / green waste and paper input may be appropriate for some soil conditioning / agricultural purposes.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Substance		Metals i	n digestate			
	mg / kg of mixed MSW digestate	g/T of mixed MSW input waste	mg / kg of VFG & paper digestate	g/T of VFG & paper input waste	Data Pedigree	
Cadmium (Cd)	2 - 4.8	1- 2.4	1 – 1.2	0.5 - 0.6	Moderate (7)	
Zinc (Zn)	533 - 1020	267 - 510	85 - 380	42.5 - 190	Moderate (7)	
Copper (Cu)	101 - 370	51- 185	14 - 88	7 - 44	Moderate (7)	
Lead (Pb)	522 - 635	261 - 318	61 - 160	31 - 80	Moderate (7)	
Nickel (Ni)	42 - 64	21 - 32	7 - 29	3.5 – 14.5	Moderate (7)	
Chromium (Cr)	87 - 292	43.5 - 146	46	23	Moderate (7)	
Mercury (Hg)	1.7 – 3.1	0.9 - 1.6	0.9	0.5	Moderate (7)	

Table 2.14 Metals in digestate from different municipal sourced feedstock processed by AD

Source AD 9 & AD 2

VFG = Vegetable, Fruit & Garden waste

2.8.6 Emissions to sewer/surface water per tonne of waste

A study for the Environment Agency (AD2) indicates that Anaerobic Digestion processes produce 100 – 330 kg of liquid per tonne of waste input. The variation is indicative of the types of AD system available (high and low solids, high and low temperature etc.). Some Anaerobic Digestion processes need water to dilute the high solids content of the waste and to recycle process bacteria. These liquids can become noxious if stored in the open and allowed to deteriorate. The liquors may be either re-circulated or removed for treatment / discharged to sewer as appropriate and depending on the system. If treated or if of a suitable quality, these liquids may be used a fertilisers (e.g. the Biorek process in Denmark). Some plant (e.g. the BTA process) denitrify the waste waters prior to discharge to sewer.

The data in Table 2.15 uses the composition of the liquors at the Water Press (AD5).

Emissions to Water / treatment	Emission waste proc	s per tonne of MS cessed g/T, where	N residual appropriate	Data
processes	Minimum	'Best Estimate'	Maximum	Pealgree
Dissolved solids	62	80	98	Moderate (8)
Total Nitrogen	7.3	10	13	Moderate (8)
Ammoniacal nitrogen	5.4	7.3	9.1	Moderate (8)
Nitrite nitrogen	No data	0.04	No data	Moderate (8)
Nitrate nitrogen	No data	0.04	No data	Moderate (8)
COD	73	100	127	Moderate (8)
BOD	18	25	33	Moderate (8)

Table 2.15	Emissions to water	/ treatment fron	AD processes

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.8.7 Emissions under non-standard conditions

The key issues for anaerobic digestion are ensuring complete digestion of wastes; effective combustion of biogas, and providing appropriate routes for disposal of digested materials. Incompletely digested material is likely to be strongly odorous. If the digestate product is not of adequate quality for spreading to land, it may need to be disposed of at landfill. Controls on the landfill should ensure that this in itself would not result in a significant environmental emission.

If biogas is not effectively combusted, then it could be released directly to air. Under these circumstances, emissions are likely to be most closely represented by fugitive gas emissions from landfill (Component 1). However, anaerobic digestion systems are normally equipped with a flexible containment system, allowing gas to be stored during any period of repair or servicing of combustion plant.

Anaerobic digestion of MSW would be regulated by the Environment Agency, with the aim of minimising the occurrence of such incidents.

Reference No	Reference Document
AD 1	Brummeler, 2000. Full Scale experience with the Biocel process. Water Science & Technology14 (3); IWA.
AD 2	Environment Agency, 2002f. Waste Pre-Treatment: A Review Report P1-344.
AD 3	National Society for Clean Air & the Environment (NSCA), 2002. Comparison of Emissions from Waste Management Options
AD 4	Wheeler P.A. and Bardos R.P., 1992 Preliminary Assessment of Heavy Metals in Anaerobic Digestion of Municipal Solid Waste
AD 5	Environment Agency, 2000a. Life Cycle Inventory Development for Waste Management Operations: Composting and Anaerobic Digestion, R&D Project Record P1/392/4
AD 6	IWM Anaerobic Digestion Working Group (1998) Anaerobic Digestion A detailed report on the latest methods and technology for the anaerobic digestion of municipal solid waste
AD 7	IEA Bioenergy, 1996, From Municipal Solid Waste Overview of Systems and Markets for Anaerobic Digestion of MSW
AD 8	Enviros, 2002. Waste Technology Options – A review for Norfolk
AD 9	CIWM, 2003. Biological Techniques in Solid Waste Management and Land Remediation
AD 10	Environment Agency, 2003g. Monitoring of Particulate Matter in Ambient Air around Waste Facilities Technical Guidance Document M17
AD 11	McLanaghan, S, 2002. Delivering the Landfill Directive: The role of new and emerging technologies.
AD 12	Alternative Waste Management Technologies and Practices Inquiry, 2000. Report of the Alternative Waste Management Technologies and Practices Inquiry. The State Government of New South Wales.
AD 13	European Commission, 2001. Waste Management Options & Climate Change.

2.8.8 Data sources

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.9 Gasification/Pyrolysis with Energy Recovery

2.9.1 Introduction

Pyrolysis and gasification, and hybrids thereof, are referred to as advanced thermal treatments. Interest in gasification and pyrolysis has heightened over the past three years and there is considerable anticipation that these technologies will prove to be viable alternatives for dealing with residual fractions municipal waste to assist in compliance with the Landfill diversion elements of the EC Landfill Directive. Pyrolysis is the thermal degradation of waste in the absence of air to produce gas (often termed syngas), liquid (pyrolysis oil) or solid (char, mainly ash and carbon). Pyrolysis generally takes place between 400-1000°C. The solid fraction may be subsequently fed into a gasification process.

Gasification takes place at higher temperatures than pyrolysis (1,000-1,400°C) in a controlled amount of oxygen. The majority of the carbon content in the waste is converted into a gaseous form (syngas). For most waste feedstock, the gas produced will contain toxic and corrosive reduced species. The gas may therefore require cleaning before combustion and post combustion there will be a Flue Gas Treatment process. The gases are burnt to produce heat, which is usually used to produce electricity. The char from either system can be further reacted with steam to produce syngas, and residual ash can be re-used or sent to landfill.

These systems are less robust for dealing with 'raw' municipal solid waste (MSW) than the established moving grate energy from waste technology, and tend to require front end treatment prior to processing in pyrolysis and gasification plant. Some of technologies have been promoted for use in processing Refuse Derived Fuel (RDF) or residues from Mechanical Biological Treatment (MBT) systems which is more homogenous than raw MSW.

2.9.2 Sources of data; limitations; assumptions; areas of uncertainty

The Environment Agency prepared a review of pyrolysis and gasification processes (PG1). The review includes a summary of release concentrations from pyrolysis and gasification systems. The report notes that there is very little published data on emissions from pyrolysis and gasification systems and that there are limitations associated with the data that is available:

- In many cases the published data is the result of a single test on a pilot or demonstration plant;
- Much of the data may have resulted from trials under conditions that may be different to, or do not reflect the full range of, those that will be experienced in reality;
- Sometimes the data may be a prediction but not clearly stated as such;
- Sometimes data is taken from different sample sets to produce a complete compositional analysis
- The averaging period for emissions to air data is rarely stated;
- It is not always clear whether the emissions data have been standardised;
- It is not always clear what waste feedstock the data relates to;



Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

• The solid residues in different processes are treated very differently making end impacts to land almost entirely dependent on the manufacturer of the system and prevailing market conditions.

The latest information from existing operational facilities has been sought to substantiate published data (PG2). This information goes some way to dealing with the concerns highlighted in the Environment Agency review.

Gaps remain in terms of the compositions of the solid residues from the processes. The diversity of products and residues from the different systems make a comparable range of compositional data from the limited dataset available impractical at this time. The lack of operational experience on MSW and the diversity of systems available is the key reason for this.

Emissions information has been provided from four operational facilites handling municipal solid waste. One facility is based in the UK, one in Germany, and two in Australia. These data have been used in conjunction with estimates of gas volumes produced per tonne of waste to derive mass release rates per tonne of waste. Gas production was estimated from data for proposed and operational plants in the UK.

The likely scale of emissions during abnormal operating conditions is similar to that described below for large-scale incineration. The available information with regard to incineration suggests that a small number of exceedances may occur annually. These incidents would not be expected to give rise to significant adverse environmental or health effects.

2.9.3 UK emissions from pyrolysis and gasification

At present there are no full scale Pyrolysis and Gasification plant operating in the UK and those on trial or small scale operations (e.g. Compact Power in Avonmouth at 4,000 Tpa) are not processing MSW except on a trial, periodic basis. Therefore the current releases to air, land and water from these systems in the UK is negligible for MSW.

2.9.4 Emissions to air per tonne of waste

The data in Table 2.16 sets out the measured emissions to air from two pyrolysis/ gasification processes, and also the areas where data are currently lacking (PG3). The median measured release value is provided, together with an indication of the range of measured emissions values.

Emissions from other processes will be dependent on the choice of combustion plant for the syngas produced in the gasification stage.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.16	Emissions to air from pyrolysis/gasification
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Substance	Emissio F	ons per tonne o processed (g/T	of waste)	Data Pedigree		
	Minimum	Best estimate	Maximum			
Nitrogen oxides	390	780	1600	Moderate (8)		
Particulates	6	12	24	Moderate (8)		
Sulphur dioxide	9	52	312	Moderate (8)		
Hydrogen chloride	16	32	64	Moderate (8)		
Hydrogen fluoride	0.11	0.34	1.0	Moderate (8)		
VOCs	3	11	44	Moderate (8)		
1,1-Dichloroethane	Not	likely to be em				
Chloroethane	Not	likely to be em	itted			
Chloroethene	Not	likely to be em	itted			
Chlorobenzene	Not	likely to be em	itted			
Tetrachloroethene	Not	likely to be em	itted			
Methane	Not	likely to be em	itted			
Cadmium	0.0017	0.0069	0.0276	Moderate (8)		
Nickel	0.02	0.040	0.08	Moderate (8)		
Arsenic	0.055	0.060	0.066	Moderate (8)		
Mercury	0.017	0.069	0.276	Moderate (8)		
Dioxins and furans (ngTEQ/T)	4 ×10 ⁻⁹	5 ×10 ⁻⁸	6×10^{-7}	Moderate (8)		
Polychlorinated biphenyls		No data				
Carbon Dioxide	No data					
Carbon Monoxide	20	100	500	Moderate (8)		

Offset to emissions from electricity generation

Pyroloysis/Gasification would normally be operated with energy recovery, resulting in the generation of electricity for export to the National Grid. Generating electricity in this way may be viewed as reducing the need to generate electricity from other sources, with a benefit in reductions in emissions from these sources. The estimated reduction in emissions as a result of avoided use of fuels is set out in Appendix 3. This suggests that there would be a net reduction in emissions to air of sulphur dioxide and particulates from the generation of electricity by the use of pyrolysis/gasification of MSW, compared to emissions from UK power generation in 2001. Emissions of oxides of nitrogen, VOCs and dioxins from pyrolysis/gasification are similar to those from power generation in 2001. Emissions of metals are similar to or higher than those from power generation in 2001.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.9.5 Emissions to land/groundwater per tonne of waste

The principal emission to land from Pyrolysis and Gasification systems is from unused solid residues of the process sent to landfill. Some systems purport to produce 100% usable residues from the system and so in theory would have little or no impacts on land provided there were markets for the residue fractions from the process and its reuse / recycling application were environmentally benign. Clearly there are however, potential environmental impacts from recycling applications such as leaching of residues used in construction applications etc. The lack of operational practice on MSW from these processes in the UK and overseas makes a realistic projection of the likely fate of solid residues from these processes unviable at this time.

Overall quantities of residue are, again, many and varied depending on the configuration and technology employed. Examples of levels of residue production are included in Table 2.17 below. It should be noted that each process envisages some reuse of residue, e.g. carbon black from gasification, 'inert' products used in construction, air pollution control residues treated and reused in chemical applications. Some processes (such as Plasma Pyrolysis) vitrify the solid residues thereby rendering them less likely to leach and whilst this will require an energy input it will reduce the environmental impact associated with disposal of the residue. An example of a leaching test of a vitrified product is included in Table 2.18, as vitrified ashes could be used directly, for example in construction operations, rather than being disposes to landfill.

Process	Observed solid residue range, kg / T processed waste input	Data Pedigree
Gasification	~170 - 300	Moderate (5)
Gasification & Pyrolysis	50 – 200	Moderate (5)
Pyrolysis	~ 300	Moderate (5)
Air Pollution Control system (required by all processes)	~20 (varies according to air pollution control system employed, would require disposal to hazardous waste landfill or additional treatment for recycling applications)	Moderate (6)

Table 2.17 Quantities of solid residues

Source: PG2, PG4 & PG5

Residues from the Air Pollution Control system will either require further processing or disposal to a landfill licensed to accept hazardous waste. The quantities and composition of residue will depend on the type of treatment system employed and the type of waste processed.

Introduction Re on		Review on Emis	Review of Information on Emissions Resea		v of Quantific niological Consequ rch Emission		tion of Health nces of	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.18	Leachable elements	of	vitrified	residue	from	plasma	pyrolysis	process	of	MSW
	derived fuel					-		-		

Leachable emissions	Emissions	s per tonne of wast	e processed (g/T)	Data
from vitrified residue	Minimum	'Best Estimate'	Maximum	Pedigree
Aluminium [7429-90-5]	Not known	<0.578	Not known	Moderate (5)
Chloride	Not known	1.428	Not known	Moderate (5)
Sulphate	Not known	<1.7	Not known	Moderate (5)
Fluoride	Not known	0.765	Not known	Moderate (5)
Cadmium	Not known	<0.0544	Not known	Moderate (5)
Mercury	Not known	<0.00204	Not known	Moderate (5)
Tin	Not known	<0.731	Not known	Moderate (5)
Lead	Not known	<0.170	Not known	Moderate (5)
Zinc	Not known	<0.0221	Not known	Moderate (5)
Nickel	Not known	<0.1071	Not known	Moderate (5)
Chromium (total)	Not known	<0.085	Not known	Moderate (5)
Copper	Not known	0.0697	Not known	Moderate (5)
Phenol	Not known	<0.0017	Not known	Moderate (5)
Iron	Not known	<0.102	Not known	Moderate (5)
Nitrate (N)	Not known	0.00068	Not known	Moderate (5)
Nitrate (N0 ₃)	Not known	0.306	Not known	Moderate (5)
Nitrite (N)	Not known	0.0068	Not known	Moderate (5)
Nitrite (NO ₂)	Not known	0.0221	Not known	Moderate (5)

61

Source: PG6

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Pyrolysis and / or gasification systems will produce a slag or char residue which may be disposed of to an appropriately licensed landfill. The data in Table 2.18 is a leachability study for a vitrified residue whilst the data included in Table 2.19 is derived from composition studies of a non vitrified residue. This residue is sent for disposal to landfill and the data is derived from a limited dataset available on MSW (Source: PG2)

Emissions to	Emissions	per tonne of waste (g/T)	e processed	Data
Landini	Minimum	'Best Estimate'	Maximum	Pedigree
Aluminium	15,000	18,000	20,600	Moderate (5)
Iron	5,040	9,700	14,300	Moderate (5)
Chloride	3,300	4,800	6,300	Moderate (5)
Sulphate	4,410	5,090	5,760	Moderate (5)
Ammonia	0.6	0.75	0.9	Moderate (5)
Cadmium	2.1	3.8	5.4	Moderate (5)
Mercury	Not known	Not known	0.06	Poor (4)
Thallium	Not known	Not known	1.2	Poor (4)
Lead	200	670	1,140	Moderate (5)
Zinc	200	1,700	3,180	Moderate (5)
Nickel	5	36	66	Moderate (5)
Chrome (total)	60	270	470	Moderate (5)
Copper	25	360	690	Moderate (5)
Phenol	0.018	0.06	0.11	Moderate (5)

 Table 2.19
 Composition data for a slag residue from a pyrolysis process

Source : PG2

2.9.6 Emissions to sewer/surface water per tonne of waste

The variety of systems which fall under the category of Pyrolysis and Gasification include many systems which do not produce effluent as it is reused as part of the process. Data on this area is unavailable because of the variety of systems and the lack of operating experience on MSW at commercial scale. Those manufacturers operating on or marketed towards MSW feedstock that were investigated (PG2) did not produce an effluent from the process. The Environment Agency (PG1) identified that little work has been published in this area, but that any liquid residues from the primary reactor or fuel gas cleaning will have high organic loadings (polycyclic aromatic hydrocarbons (PAHs), and phenols). The composition and quantity of effluent produced by the Air Pollution Control system will depend on the type of system used. One of the operational facilities investigated uses a wet scrubbing air pollution control system which produces 6 litres of waste water per tonne of waste input to the plant. This effluent is rich in Alkali salts, in particular Ammonia Chloride, and requires further treatment before release or binding prior to disposal at an appropriately licensed landfill.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.9.7 Data sources

Reference No.	Reference document
PG1	Environment Agency, 2001h. Review of BAT for New Waste Incineration Issues: Part 1 Waste Pyrolysis & Gasification Activities, R&D Technical Report P4-100TR
PG2	Industry Sources and Information from the following operators & system manufacturers: Wastegen, Brightstar, Compact Power, Thermoselect, Global Olivine
PG3	National Society for Clean Air & the Environment (NSCA), 2002. Comparison of Emissions from Waste Management Options
PG4	C-Tech Innovation Ltd, 2003. Thermal Methods of Municipal Waste Treatment
PG5	McFarlane K. et al. 1997. An Assessment of UK Systems for the Thermal Conversion of Waste
PG6	Lapa et al., 2002. An ecotoxic risk assessment of residue materials produced by the plasma pyrolysis/vitrification (PP/V) Process Waste Management 22.
PG 7	Environment Agency, 2002f. Waste Pre-Treatment: A Review Report P1-344.
PG 8	Environment Agency, 2003g. Monitoring of Particulate Matter in Ambient Air around Waste Facilities, draft Technical Guidance Document M17

63

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

2.10 Unsegregated incineration with energy recovery

2.10.1 Introduction

Incineration involves the controlled combustion of waste at temperatures above 850°C. This study considers mass burn incineration, with energy recovery. Traditional fuels (e.g. fuel oil or gas) are only used during start-up, and therefore the majority of emissions from the incinerator plant are due to waste combustion.

The enforcement of a number of European Directive limits over recent years has drastically reduced the concentration of many pollutants in emissions to air from incinerators. The Directive on the Incineration of Waste (European Commission, 2000) imposes even stricter emission limits. New plant was required to comply with these limits by the end of 2002, and existing plant will need to comply by the end of 2007. In fact, for the majority of substances, MSW incinerator processes already comply with the waste incineration directive.

2.10.2 Sources of data; limitations; assumptions; areas of uncertainty

The assessment of emissions resulting from incineration falls into the following elements:

- Assessment of emissions from existing operational incinerators based on operational data;
- Assessment of emissions from a selection of existing incinerators, based on the assumption that emissions from these plants comply with limits specified in the Waste Incineration Directive;
- Assessment of historical emissions from incinerators i.e before the Waste Incineration Directive and before the Municipal Waste Incineration Directive. This information is based on data from the National Atmospheric Emissions Inventory (NAEI).

The assessment of emissions to air from incineration of MSW focuses on emissions through the incinerator waste gas stack. No assessment is made of emissions created by the storage and processing of waste prior to incineration, nor of potential releases of ash to air. Emissions to water, disposal to landfill and reuse have been calculated, on the basis of Environment Agency Pollution Inventory (PI) data alone.

Methodology

Where emissions to air from a plant have been derived from monitoring data the following equation has been used:

Emissions of a substance per tonne of waste (grams/tonne) = Emission concentration of a substance in efflux gas (grams/metre³)

Volumetric flow rate of
 efflux gas (metre³/hour)

Waste throughput (tonne/hour)
Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

To carry on this calculation, concentrations and flow rates must be specified at the same conditions i.e. normalised or stack release conditions. All data in this section has also been referenced to standard conditions: Temperature: 273°K; Moisture Content: 0%; Oxygen Content: 11%; Pressure: 101.3 kPa.

Where possible, waste throughput data has been based on actual figures supplied by the plant operator. Where data was not available, it has been assumed that the plant was operating at design capacity. Comparing the capacity of the incineration plant with the known amount of waste incinerated in 2000 indicates that this is a reasonable assumption.

In the case of monitoring data which is below the limit of detection (LOD), or Pollution Inventory data which is below the reporting threshold (BRT), the value taken has been assumed to be half that of the limit, e.g. a value of <10 tonnes is assumed to be 5 tonnes \pm 5 tonnes.

The estimated total releases have been derived by multiplying the best estimate of release rates for a pollutant by the known throughput of MSW for 2000.

Source data

The data used in this report have been drawn from a variety of sources. These include raw data as submitted to the Environment Agency (EA) by operators, along with reviews of other research on the subject, as explained below.

The PI database held by the Environment Agency contains the total mass of a particular substance emitted in a given year, and is based on actual emissions measurements.

For the majority of plant the annual waste throughput provided by operators has been used in conjunction with the PI data to calculate the average mass of a substance emitted per tonne of MSW combusted, over the course of a typical operational year. Where the annual throughput data was not available, the throughput has been assumed to be equal to the design capacity of the plant.

PI data has been used for eleven mass burn incinerators using energy recovery which were operating in 2000 in England, with the exception of Bolton Incinerator. Bolton Incinerator was under going commissioning during the first four months of 2000, and as such the emissions from the plant over the course of the year do not represent typical operation. Data from the Bolton incinerator have therefore not been included in this study.

Emissions measurements recorded by the operator, and in one case by the EA, have been supplied for three plants for the year 2001. This data has been used in conjunction with the PI data for 2000. It has been assumed that emissions are representative and consistent over the year. The median values have been taken for emissions data sets, after scanning and removal of data considered likely to be unreliable.

Following recent upgrades, the eleven UK MSW incinerators considered in this chapter are equipped with a similar range of abatement (Environmental Services Association, 2003):

Teesside: Gas Scrubber; Bag-house filter

Coventry: Dry Scrubbing; Fabric Filter

Introduction		Review on Emis	of Information ssions	formation Review of Epidemiological Research		Quantifica Conseque Emissions	Quantification of Health Consequences of Emissions		n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & ic ues	Conclusions	
Introduction	luction Scope Sources Materials Recovery Facilities Composting (in-vessel)		Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

Nottingham: Dry injection of Lime and activated carbon; Bag filter

- SELCHP: Activated carbon; semi-dry scrubbers (lime and water mixture); Bag filter
- Stoke Spray dryer; Selective non-catalytic reduction for NOx abatement; Bag filter
- Tyseley: Gas Scrubber; Bag-house filter
- Wolverhampton: Gas Scrubber; Bag-house filter
- Dudley: Gas Scrubber; Bag-house filter
- Sheffield: Gas Scrubber; Bag-house filter

Other research on incinerator releases has been reviewed, and included in the summary table. The accuracy and reliability of reported data has not been verified, and in many cases there are no details of calculation methodologies or inherent assumptions. The input waste streams will vary from plant to plant. Also many of the reports reviewed were based on non-UK plant and data.

The Public Acceptability of Incineration (NSCA, 2001) report and the UK Emissions of Air Pollutants 1970 – 1999 (NAEI, 2001) report have been reviewed. However, data from these reports have not been included in this assessment because they are based on industry returns to the EA and the Agency's Chemical Release Inventory (CRI) which are between three and ten years old. These estimates were calculated using factors that may not be applicable to the current suite of incinerators.

All data used has been accepted de facto. Where possible methodologies used for deriving concentrations have been confirmed to be consistent, however this has not been possible in all cases.

Uncertainty

All data used have a degree of uncertainty attached, arising from measurement accuracy, uncertainty associated with the assumptions that are required during calculations, and variability between processes.

Measurement uncertainty is typically $\pm 10\%$ to $\pm 50\%$. The data used in Reference 12 were supplied with inherent error values. These were between 10 and 20% for the extractive sampling required for metals, but much less for on-line sampling techniques used for nitrogen oxides and sulphur dioxide, ranging from 1 to 10%.

The data generated by spot sampling need to be used with particular care, because the waste burnt in incinerators is inherently variable in composition and calorific value. Individual spot measurements may give data which are not representative of the long-term emissions from an incineration process. However, greater confidence can be obtained from a larger set of such measurements. In this case, an indication of the reliability of the data was obtained from the range of data and/or the standard deviation of the data set.

The second area of uncertainty relates to the waste throughput values. Emissions tests will normally be carried out when plants are operating close to, or above, their maximum rating. Information on process throughputs is not routinely recorded, and

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequence Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & ic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

so some assumptions have to be made regarding waste inputs. This could have introduced an uncertainty of the order of \pm 25%.

The estimated variability range shown in Table 2.20 and Table 2.21, has been derived by allowing for:

- Measurement uncertainty for the substance;
- Variability in the data set;
- Uncertainty in the waste throughput (± 25%)

It is inappropriate to use the mean and standard deviation of the data set to assess the uncertainty within the data, because the data may not be normally distributed. To overcome this, where there are 5 or more data points, the standard deviations of the logarithms of the measured values were used as the uncertainty range. The overall uncertainty is the root-mean-square uncertainty from the three parameters listed above.

Where there are fewer than 5 data points, the best estimate has been based on the median of the data set. The uncertainty range was calculated from the range of values, together with the other factors set out above. The overall uncertainty was based on the combination of these uncertainties on a root-mean-square basis.

The key data inputs used to evaluate data pedigree are the pollution inventory estimates, and the waste throughput rates at each facility. Of these, the greatest variability is associated with emissions concentrations measurements.

2.10.3 Emissions to air per tonne of waste

The best estimate for substances released by MSW incineration have been determined by taking the median value from the range of reliable data. The UK inventory of released substances was calculated multiplying the best estimate of the release rate by the total throughput data for the all the incinerators

Table 2.20 provides results based on operational data from existing plants, and Pl data from the Environment Agency.

Table 2.21 provides estimates of emissions for a selection of operational plants, assuming compliance with Waste Incineration Directive limits.

Introd	Introduction		Review on Emi	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		on of the tal ces of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introdu	uction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

 Table 2.20
 Emission rates to air using operational incinerator data

Substance	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Estimated Total UK Releases (2000) (T)	Data Refs.	Comments	Data Pedigree
Nitrogen Oxides	1050	1600	2400	3800	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11,12, 13, 14, 15,16,17		Good (9)
Total Particulates	11	38	130	93	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11,12, 13, 14, 15, 16, 17		Good (9)
Sulphur Oxides	16	42	110	102	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15, 16, 17		Good (9)
Hydrogen Chloride	35	58	100	140	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15, 17		Good (9)
Hydrogen Fluoride	0.3	1	3.5	2.7	1, 2, 3, 6, 7, 8, 9, 10, 11, 12, 13, 17		Good (9)
Volatile Organic Compounds	2	8	30	18	1, 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 13, 14, 16, 17	6 of 11 ISR data points are BRT	Moderate (8)
1,1 – Dichloroethane						No data	
Chloroethane						No data	
Chloroethene		0.1		0.3	6	Data below LOD	Moderate (7)
Chlorobenzene						No data	
Tetrachloroethe ne		0.1		0.3	6	Data below LOD	Moderate (7)
Methane	9	19	40	45	3, 6, 7, 9	Figure based on ISR data which is all BRT	Moderate (7)
Cadmium	0.001	0.005	0.02	0.01	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12,14, 17		Good (9)
Nickel	0.02	0.05	0.15	0.1	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 14	7 of 10 ISR data points are below LOD	Moderate (8)
Arsenic	0.001	0.005	0.03	0.01	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11,12,14	7 of 10 ISR data points are BRT	Moderate (8)
Mercury	0.01	0.05	0.3	0.1	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 17		Moderate (8)
Dioxins and Furans	100 ng TEQ/T	400 ng TEQ/T	1600 ng TEQ/T	1 x 10 ⁻⁷	1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 16, 17		Good (9)
Dioxin-like Polychlorinated Biphenyls		0.1 mg TEQ/T		3 x 10 ⁻⁵	6	Data below LOD	Moderate (7)
Carbon Dioxide	700000	1000000	1400000	2310000	1, 2, 5, 6, 7, 8, 9, 10, 11, 17		Good (9)

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		on of the tal ces of	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.21	Emission	rates	to	air	assuming	incinerators	operate	at	Waste	Incineration
	Directive I	imits								

Substance	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Data Pedigree
Nitrogen Oxides	600	1100	1600	Very good (13)
Total Particulates	40	60	80	Very good (13)
Sulphur Oxides	180	280	380	Very good (13)
Hydrogen Chloride	40	60	80	Very good (13)
Hydrogen Fluoride	4	6	8	Very good (13)
Volatile Organic Compounds	40	60	80	Very good (13)
1,1 –Dichloroethane		No Limit		
Chloroethane		No Limit		
Chloroethene		No Limit		
Chlorobenzene		No Limit		
Tetrachloroethene		No Limit		
Methane		No Limit		
Cadmium	0.2	0.3	0.4	Very good (13)
Nickel	0.3	0.6	0.9	Very good (13)
Arsenic	0.3	0.6	0.9	Very good (13)
Mercury	0.4	0.6	0.8	Very good (13)
Dioxins and Furans	540 ng TEQ / T	560 ng TEQ / T	580 ng TEQ / T	Very good (13)
Dioxin-like polychlorinated Biphenyls		No Limit		
Carbon Dioxide		No Limit		

Note: The best estimate for emissions from plants assuming WID release rates has been calculated using the mean of the data set.

Offset to emissions from electricity generation

Incineration is normally operated with energy recovery, resulting in the generation of electricity for export to the National Grid. Generating electricity in this way may be viewed as reducing the need to generate electricity from other sources, with a benefit in reductions in emissions from these sources. The estimated reduction in emissions as a result of avoided use of fuels is set out in Appendix 3. This suggests that there is a net reduction in emissions to air of sulphur dioxide and particulates from the generation of electricity from incineration of MSW, compared to emissions from UK power generation in 2001. There is a net increase in other emissions from the generation of electricity by MSW incineration, compared to power generation in 2001.

Historical context for incinerator emissions

Emissions from incineration have varied as new legislation and technology has been implemented. Emissions to air during 1980 and 1990 have been calculated, based on total releases from UK incinerators reported by the National Atmospheric Emissions Inventory. The total mass of municipal solid waste incinerated was taken

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

from The Chartered Institute of Public Finance and Accountancy for 1980 data (2,780,000 tonnes) and from Greenpeace for 1990 (2,500,000 tonnes). The emission rates to air from incinerators during 1980, 1990 and 2000 are presented in the table below.

Substance	E	stimated emission	ons to air (g	J/T except where	stated other	wise)
Substance	1980	Data Pedigree	1990	Data Pedigree	2000	Data Pedigree
Nitrogen Oxides	1878	Moderate (5)	1580	Moderate (6)	1600	Good (9)
Total Particulates	313	Poor (4)	264	Moderate (6)	38	Good (9)
Sulphur Dioxide	1421	Moderate (5)	1196	Moderate (6)	42	Good (9)
Hydrogen Chloride	3791	Moderate (5)	20	Moderate (6)	58	Good (9)
Hydrogen Fluoride	No data		No data	Moderate (6)	1	Good (9)
Volatile Organic Compounds	25	Poor (4)	20	Moderate (5)	8	Moderate (8)
Cadmium	2.6	Poor (3)	16	Moderate (5)	0.005	Good (9)
Nickel	2.8	Poor (3)	28	Moderate (5)	0.05	Moderate (8)
Arsenic	0.40	Poor (3)	0.33	Moderate (5)	0.005	Moderate (8)
Mercury	1.8	Poor (3)	2.2	Moderate (5)	0.05	Good (9)
Dioxins and Furans	No data		0.00018 g TEQ/T	Moderate (5)	4 × 10 ⁻⁷ g TEQ/T	Good (9)
Dioxin-like polychlorinated Biphenyls	No data		0.0035 g TEQ/T	Moderate (5)	0.0001 g TEQ/T	Moderate (7)

Table 2.22 Historical context for incinerator emissions

Emissions to air of dioxins and furans, PCBs and trace metals from MSW incineration have reduced considerably between 1990 and the present day. This reduction has arisen largely as a result of increasingly stringent limits on emissions of these substances set in European directives (89/369/EEC; 89/429/EEC; 2000/76/EC). In the late 1990s, MSW incinerators were either shut down or upgraded to meet the emissions limits set out in the 1989 European directives.

To meet the emission limits for dioxins and furans and metals, most MSW incinerators now operate an air pollution control system based on the injection of materials to absorb these substances, and a filter system to remove the injected materials with the trace contaminants. One approach is to inject activated carbon powder which absorbs semi-volatile substances including mercury and dioxins and furans. The activated carbon is then trapped in bag filters, and removed as air pollution control residues for disposal, normally at a suitably licensed landfill.

Emissions from incinerators under non-standard operating conditions

As set out in Chapter 1, a detailed evaluation of operation under abnormal conditions does not form part of this study. Municipal waste incinerators operate to the increasingly stringent terms of licences (issued under the Integrated Pollution Control regime) or permits (issued under the Integrated Pollution Prevention and Control regime). From time to time, incinerators experience periods of operation outside the permitted range. Examples of where this occurs include:

• Emissions may exceed the normal operating limits during start-up and shutdown (this may well be allowed for within the terms of the operating licence).

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequence Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

- Short-term fluctuations in emissions of specific substances to air may exceed the operating limits. The occurrence of these peaks in emissions can normally be minimised by good mixing of wastes, and they can also be prevented or controlled by adjusting the use of gas cleaning systems.
- It is in principle possible for emissions of substances which are not measured continuously to exceed the prescribed limits during periods between measurements. However, it is often possible to use measurements of continuously measured substances as a proxy for substances which are not measured continuously for example, hydrogen chloride can be used as a proxy for dioxins and furans, or particulates can be used as a proxy for metals. Controlling the measured emissions reduces the likelihood of other substances being emitted at excessive concentrations.

As well as this, the Environment Agency carries out a programme of random, unannounced emissions monitoring checks on all its regulated processes. Systems are also now available for continuous sampling of flue gases over a period of weeks or months. This sample can then be analysed to determine the average level of emissions of dioxins and furans over this period. Tests carried out in this way at a plant in Oostende, Belgium, demonstrated that emissions of dioxins and furans were in compliance with the Waste Incineration Directive limits by a considerable margin.

Environment Agency records indicate that 56 incidents of emissions outside permitted limits occurred at the 14 incinerators accepting MSW in the UK in 2003. This corresponds to four incidents per incinerator on average, although the numbers vary from 0 to 15 at different facilities. The highest number of incidents occurred at a new incinerator which was being commissioned. Three quarters of the incidents related to increased emissions of carbon monoxide and hydrogen chloride, which would not be expected to result in any significant environmental effects. There were four incidents of dioxins and furans above permitted levels, and one incident of cadmium emissions above permitted levels.

Any emission above prescribed limits is of concern, and it is important that these incidents are investigated and their recurrence prevented. However, the low frequency of these incidents and the lack of any consistent evidence for health effects in people living near waste to energy facilities (see Chapter 3) suggest that emissions above consented limits are not a significant issue for waste incinerators. Also, an exceedance over a short period is not likely to have a significant effect on emissions averaged over a long period such as a year. Exceedances may be more likely to occur from facilities which are undergoing commissioning, and particular attention should be paid to regulation of facilities in these circumstances.

MSW incineration processes are regulated by the Environment Agency, with the aim of minimising the occurrence of such incidents.

Conclusions

The mass of each substance released per tonne of MSW incinerated are set out in Table 2.20 and 2.21.

The range associated with each substance highlights the variation in the magnitude of releases from incineration processes.

Data are particularly sparse for individual VOCs and PCBs.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequence Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Comparison of the best estimates for operational data and WID limits suggests that for the majority of substances, existing operational plants already comply with the limits in the WID. This suggests that when the WID limits come into force on existing plant the improvement in emissions per tonne of MSW will not be substantial. The single exception to this is likely to be nitrogen oxides.

Most information sources on incineration do not reference biohazards. This reflects the main focus of attention on emissions from the main stack. Crook et al (1987), on behalf of the Health and Safety Executive, examined bioaerosols in two incinerators as well as at other waste disposal locations. This study, together with that of Rahkonen, 1992 measured concentrations of biohazards within the facility or immediately outside. It was found that levels reduced to background levels within 50 metres of the sites, suggesting that emissions are not likely to be significant. Levels are highest within the facility itself.

2.10.4 Incineration outside the UK

The information reviewed on non-UK incinerators has primarily concentrated on the European Union, the USA, Canada and Japan as these nations have a similar level of technological advancement as the UK. Throughout the European Union the emissions and impacts from incinerators are largely comparable to those in the UK. This is primarily due to the Europe-wide transposition of EU legislative controls, for example the Waste Incineration Directive, into national regulations.

The proportion of waste disposal to incinerators in Europe, USA and Japan varies greatly. In the late 1990s, MSW incinerators in Europe were either closed or upgraded to meet emissions limits set out in the Municipal Waste Incineration Directives (89/369/EEC and 89/429/EEC). Table 2.23 sets out the quantities of municipal solid waste collected, landfilled and incinerated in each EU country, Iceland, Norway the USA and Japan. The figures are expressed as kg per person. Data is for the year 2000 unless otherwise indicated (European Commission, 2003).

Introduction		Review on Emis	Review of Information on Emissions		Review of Qu Epidemiological Co Research Er		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.23	Comparison of the quantities of MSW collected, landfilled and incinerated (kg per
	person)

Country	MSW collected	MSW incinerated	MSW landfilled	% incinerated
Denmark	665	347	67	52%
Luxembourg	648 (1999 data)	310 (1999 data)	140 (1999 data)	48%
Sweden	429	165	138	39%
Netherlands	615	233	87	38%
Belgium	484	163	134	34%
France	531	176	244	33%
Germany	555	128	182 (1999 data)	23%
Japan	407 (1999 data)	86 (1999 data)	27 (1999 data)	21%
Portugal	444	91 (1999 data)	334	21%
USA	749	114 (1999 data)	441 (1999 data)	15%
Norway	615	90	336	15%
Finland	483	52	306	11%
Austria	556 (1999 data)	56 (1999 data)	192 (1999 data)	10%
United Kingdom	558 (1999 data)	49 (1999 data)	511 (1999 data)	9%
Iceland	710	61	555	9%
Spain	520	43	319	8%
Italy	502	37 (1999 data)	377 (1999 data)	7%
Ireland	626		554	0%
EU 15	535	105	291	20%

Note: some anomalies occur due to rounding errors. UK data for 2001/02 quoted in Chapter 1 indicates MSW collected: 497 kg per person; MSW incinerated: 45 kg per person; MSW landfilled: 382 kg per person.

Table 2.23 illustrates that across the EU, USA and Japan the percentages of MSW incinerated varies greatly, ranging from 52% in Denmark to 0% in Ireland.

Across Europe in recent years, as emission controls for incinerators have become progressively more stringent, smaller scale incinerators are being phased out where it is not economically viable to install the technical upgrades required to achieve the revised emissions limits. This incinerator capacity is being replaced by fewer, but larger, more advanced incinerators capable of achieving the new emission limits. The net impact of these new incinerators is likely to be lower than the impact of the older more polluting incinerators, although this will be offset to some degree by the impacts of greater distances travelled for collection and disposal of waste.

Despite the large difference between the percentage of MSW incinerated in the UK, compared to some other EU countries, the technological and emissions levels of current incineration plants is similar, primarily due to the EU-wide legislative requirements. Similar legislative requirements in the USA and Canada also ensure that the emissions and technology is comparable to the UK situation.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & ic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.10.5 Solid residues

Incineration of waste produces two main forms of solid product: air pollution control (APC) residue (also known as fly ash) and boiler bottom ash. Bottom ash originates from the incinerator grate and boiler whereas air pollution control residues are collected from the air pollution control system, for example filter bags.

Bottom ash is often reused. Where it is not reused it is disposed of to landfill as non-special waste. Air pollution control residues are normally disposed of to landfill as special waste.

Disposal rates of incinerator ash to landfill have been derived from Pollution Inventory data for 2000. Data from 8 of the 10 fully operational UK incinerators in 2000 have been used. Data from the remaining two incinerators were disregarded, as the data appeared to have been misclassified or wrongly reported in some other way.

Residue	Lower estimate (T residue / T waste processed)	Best estimate (T residue / T waste processed)	Upper estimate (T residue / T waste processed)	Estimated Total UK Releases from Waste Processed during 2000 (kT)	Data Refs.	Data Pedigree
Re-use		0.068		162	1, 2, 3, 4, 6, 7, 8, 9, 10, 11	Good (9)
Non-special waste	0.11	0.18	0.30	443	1, 2, 4, 6, 7, 8, 9, 10	Good (9)
Special waste	0.02	0.03	0.05	82	1, 2, 4, 6, 7, 8, 9, 10	Good (9)
Total waste to landfill	0.13	0.22	0.35	525		Good (9)

Table 2.24 Solid residues from MSW incineration	Table 2.24	Solid residues	from	MSW	incineration
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The uncertainty range in total wastes going to landfill reflects the fact that the majority of solid residues are re-used. Relatively small differences in the proportion of residues being re-used (e.g. 70% compared to 80% of bottom ash) would have a more significant influence on the amount of waste going to landfill (e.g. 30% compared to 20% of bottom ash – one and a half times as much).

Introduction		Review on Emi	Review of Information on Emissions		Review of Qua Epidemiological Con Research Emi		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.25 Production rates of substances in bottom ash (non-special waste)

Substance	Lower estimate (g/T waste processed)	Best estimate (g/T waste processed)	Upper estimate (g/T waste processed)	Estimated Total UK Releases from Waste Processed during 2000 (T)	Data Pedigree
Dioxins and Furans (as TEQ)	$5.5 imes 10^{-7}$	$9.7 imes10^{-6}$	$5.9 imes10^{-5}$	$2.4 imes10^{-5}$	Good (9)
Aluminium	2,000	5,400	15,000	13,000	Good (9)
Arsenic	1.8	4.8	13	12	Good (9)
Barium [7440-39-3]	89	240	650	580	Good (9)
Calcium [7440-70-2]	7,800	21,000	57,000	51,000	Good (9)
Cadmium	2.8	7.6	21	19	Good (9)
Cobalt	1.2	3.3	8.8	7.9	Good (9)
Chromium	21	56	150	140	Good (9)
Copper	150	390	1,100	950	Good (9)
Iron	2,200	5,900	16,000	14,000	Good (9)
Mercury	0.026	0.070	0.19	0.17	Good (9)
Potassium [7440-09-7]	510	1,400	3,700	3,300	Good (9)
Magnesium	760	2,000	5,500	5,000	Good (9)
Manganese	83	220	600	540	Good (9)
Sodium [7440-23-5]	1,600	4,300	12,000	10,000	Good (9)
Nickel	5.3	14	39	35	Good (9)
Lead	180	480	1,300	1,200	Good (9)
Antimony	11	30	80	72	Good (9)
Tin	120	320	850	770	Good (9)
Titanium [7440-32-6]	260	690	1,900	1,700	Good (9)
Vanadium	5.2	14	38	34	Good (9)
Zinc	320	850	2,300	2,100	Good (9)
Carbonate [3812-32-6]	660	1,800	4,800	4,300	Good (9)
Fluoride	13	36	96	87	Good (9)
Chlorine	290	790	2,100	1,900	Good (9)
Sulphite [14265-45-3]	15	42	110	100	Good (9)
Sulphate [14808-79-8]	1,540	4,200	11,000	10,000	Good (9)

Note: The emission rates have been calculated from data in Ref 21. The estimated total releases have been derived from the total MSW throughput for UK incinerators in 2000 of 2400 kilotonnes.

Introduction		Review on Emis	Review of Information on Emissions		Review of Qu Epidemiological Co Research Er		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

 Table 2.26
 Production rates of substances in air pollution control residues (special waste)

Substance	Lower estimate (g/T MSW except where stated otherwise)	Best estimate (g/T MSW except where stated otherwise)	Upper estimate (g/T MSW except where stated otherwise)	Estimated Total UK Releases from Waste Processed during 2000 (T)	Data Pedigree
тос	230	460	910	1119	Good (9)
Dioxins and Furans (as TEQ)	$1.2 imes 10^{-5}$	$2.7 imes10^{-5}$	$5.8 imes10^{-5}$	$6.5 imes10^{-5}$	Good (9)
Arsenic	0.69	1.1	1.6	2.6	Good (9)
Cadmium	2.0	2.8	3.9	6.8	Good (9)
Cobalt	0.21	0.68	2.2	1.7	Good (9)
Chromium	1.6	2.2	3.0	5.4	Good (9)
Copper	9.1	12	17	30	Good (9)
Mercury	0.18	0.25	0.35	0.61	Good (9)
Manganese	8.1	11.8	17.0	29	Good (9)
Nickel	0.61	0.81	1.1	2.0	Good (9)
Lead	46	65	93	159	Good (9)
Antimony	7.4	10	12	23	Good (9)
Tin	11	19	35	47	Good (9)
Thallium	0.19	0.31	0.50	0.74	Good (9)
Vanadium	0.41	1.6	6.3	3.9	Good (9)

Note: The emission rates have been calculated from data in Ref 22. The estimated total releases have been derived from the total MSW throughput for UK incinerators in 2000 of 2400 kilotonnes.

Bottom ash generated from incineration of MSW is commonly re-used in aggregates (e.g. road construction), bulk fill (e.g. embankments) and in building materials (e.g. concrete blocks). The estimated quantities of bottom ash reused by this method have been derived from the PI database. Within the database there are separate data and definitions relating to the categories 'reuse' and 'recovery'. However incinerator operators vary in their use of these terms. This problem is currently being addressed by the Environment Agency who are in the process of introducing new definitions for ash disposal methods.

For this current study 'reuse' has been assumed to mean bottom ash reprocessed into aggregates, bulk fill or building materials; whereas 'recovery' was assumed to mean direct recovery of energy (not included in this study) and ferrous material from the ash. Therefore we have only used PI data relating to 'reuse' of bottom ash. The PI data records the quantity of MSW handled by each incinerator and the quantity of ash sent for reuse. The composition of the ash has been derived from: Ref. 19, Ref. 20, Ref. 21, communication with WRc, and unpublished data from Huddersfield Incinerator.

Not all UK incinerators reuse ash, therefore the best estimate for substances released by MSW incineration to materials re-use has been determined by calculating the log mean value from the range of data to establish the quantity of ash sent for reuse by the five UK incinerators for which data are available. The composition of the ash is assumed to be similar for each incinerator.

Introduction		Review on Emis	Review of Information on Emissions		Review of Quantific Epidemiological Consequ Research Emission		tion of Health nces of	of Health s of Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.27 Emissions rates from ash reuse using operational incinerator data

Substance	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Estimated Total UK Releases during 2000 (T)	Data Pedigree
Dioxins and Furans (as TEQ 2,3,7,8-PCDD)	7.0 x 10 ⁻⁷	2.3 x 10 ⁻⁶	7.8 x ⁻⁶	5.61 x 10 ⁻⁶	Moderate (8)
Aluminium	414	1378	4589	3324	Good (9)
Arsenic	0	1	4	3	Good (9)
Barium	18	61	204	148	Good (9)
Calcium	1625	5409	18008	13044	Good (9)
Cadmium	1	2	7	5	Good (9)
Cobalt	<20	<20	<20	<48	Good (9)
Chromium	4	14	47	34	Good (9)
Copper	30	100	333	241	Good (9)
Iron	453	1507	5017	3634	Good (9)
Mercury	<0.1	<0.1	<0.1	<2.4	Good (9)
Potassium	106	351	1170	848	Good (9)
Magnesium	157	523	1740	1261	Good (9)
Manganese	17	57	190	138	Good (9)
Sodium	328	1092	3637	2634	Good (9)
Nickel	1	4	12	9	Good (9)
Lead	37	124	412	299	Good (9)
Antimony	2	8	25	18	Good (9)
Tin	24	81	269	195	Good (9)
Titanium	53	177	590	427	Good (9)
Vanadium	1	4	12	9	Good (9)
Zinc	66	218	726	526	Good (9)
Carbonate	136	454	1510	1094	Good (9)
Fluoride	3	9	30	22	Good (9)
Chloride	61	202	671	486	Good (9)
Sulphite	3	11	35	26	Good (9)
Sulphate	430	1420	4700	3400	Moderate (7)

The estimated total releases have been derived from the total MSW throughput for UK incinerators in 2000 of 2400 kilotonnes.

The quantity of ash generated per tonne of combusted MSW and the quantity of the ash which is subsequently sent for reuse was established from the PI database. Using the ash composition data the 'best estimate', were derived using a log mean calculation. The 'lower' and 'upper' values were derived by calculating an error based on the root mean square of the best estimates (Ref 19, Ref 20, Ref 21 and communication with WRc).

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Emissions under non-standard operating conditions

Incinerator ash streams are normally re-used or disposed of responsibly. A widely reported incident arose during which ash from the Byker incinerator in Newcastle was re-used on footpaths, including those at nearby allotments. In this case, the releases directly to land were greater than would normally occur. This has been investigated by the Environment Agency and Newcastle City Council.

In a separate incident, ash containing a mixture of bottom ash and air pollution control residues was used for manufacture of building blocks. This resulted in an increased release of these residues and their constituents to the environment (rather than to contained disposal). However, because of the stabilisation provided by the cement, and the normal use of these blocks in non-aggressive environments, an investigation by the Environment Agency found that there were no significant health risks associated with this incident. As noted above, MSW incineration processes are regulated by the Environment Agency, with the aim of minimising the occurrence of such incidents.

2.10.6 Emissions to sewer/surface water per tonne of waste

Most incinerators now have anhydrous (i.e. dry or semi-dry) gas cleaning systems for flue gases which result in no emissions to water from air pollution abatement systems.

Emissions to sewer and surface water are generated from quenching water used in the ash pits. Of the ten incinerators assessed in this report nine report emissions to the sewerage system.

The substances to be assessed have been selected from the list of substances released to water that are regulated according to the WID.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		on of the tal ces of	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Introduction	Introduction Scope Sources Materials Recovery Facilities Cr		Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

Table 2.28 Emissions rates to sewer (g of emission per tonne of MSW processed) using operational incinerator data

Substance	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Estimated Total UK Releases from Waste Processed during 2000 (T)	Data Refs.	Comments	Data Pedigree
Total Suspended Solids						No data	
Mercury	0.0002	0.0003	0.0006	0.001	3, 6, 7, 8, 9, 10, 11	6 of 7 ISR data points are BRT	Good (10)
Cadmium	0.002	0.003	0.006	0.01	2, 3, 6, 7, 8, 9, 10, 11	7 of 8 ISR data points are BRT	Good (10)
Thallium						No data	
Arsenic	0.006	0.01	0.030	0.03	3, 6, 7, 8, 9, 10	All 6 ISR data points are BRT	Good (10)
Lead	0.05	0.08	0.15	0.2	3, 4, 6, 7, 8, 9, 10, 11	7 of 8 ISR data points are BRT	Good (10)
Chromium	0.04	0.07	0.15	0.2	3, 4, 6, 7, 8, 9, 10, 11	7 of 8 ISR data points are BRT	Good (10)
Copper	0.04	0.07	0.15	0.2	3, 4, 6, 7, 8, 9, 10, 11	7 of 8 ISR data points are BRT	Good (10)
Nickel	0.04	0.07	0.15	0.2	3, 4, 6, 7, 8, 9, 10, 11	7 of 8 ISR data points are BRT	Good (10)
Zinc	0.02	0.07	0.2	0.2	3, <mark>4, 6, 7, 8</mark> , 9, 11	6 of 7 ISR data points are BRT	Good (10)
Dioxins and Furans						No data	

The estimated total releases have been derived from the total MSW throughput for UK incinerators in 2000 of 2400 kilotonnes.

Only one of the UK incinerators assessed reports emissions to controlled waters. Data for the release rates of effluent to controlled waters from this incinerator in 2000 is presented in the table below.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction Scope Sources		Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

 Table 2.29
 Emissions from one incinerator to controlled waters

Substance	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Comments	Data Pedigree
Total Suspended Solids		No data			
Mercury	0.00010	0.00012	0.00015	BRT	Good (9)
Cadmium	0.0008	0.0009	0.0012	BRT	Good (9)
Thallium		No data			
Arsenic	0.0044	0.0055	0.0069	BRT	Good (9)
Lead	0.015	0.018	0.023	BRT	Good (9)
Chromium	0.015	0.018	0.023	BRT	Good (9)
Copper	0.015	0.018	0.023	BRT	Good (9)
Nickel	0.015	0.018	0.023	BRT	Good (9)
Zinc	0.18	0.21	0.27		Good (9)
Dioxins and Furans		No data			

Data sources -incineration

Reference No.	Reference source
1	Teesside Incinerator, Cleveland – Pollution Inventory (PI) 2000. (235,000 tonnes in 2000)
2	Coventry Incinerator – PI 2000. Assumed to have operated at design capacity in 2000. (215,000 tonnes per year)
3	Edmonton Incinerator – PI 2000. (542,000 tonnes in 2000) The data for carbon dioxide has been excluded from Table 2.20 and subsequent evaluation because it is suspect.
4	Eastcroft Incinerator, Nottingham – PI 2000. (150,000 tonnes in 2000)
5	Eastcroft Incinerator, Nottingham – PI 2001. The incinerator has been assumed to operate at design capacity. (150,000 tonnes per year).
6	SELCHP Incinerator, Lewisham – PI 2000. (416,000 tonnes in 2000).
7	Stoke Incinerator, Stoke-on-Trent – PI 2000. (200,000 tonnes in 2000).
8	Tyseley Incinerator, Birmingham – PI 2000. (325,000 tonnes in 2000).
9	Wolverhampton Incinerator – PI 2000. (116,000 tonnes in 2000).
10	Dudley Incinerator, Birmingham – PI 2000. (99,000 tonnes in 2000).
11	Sheffield Incinerator – PI 2000. (98,000 tonnes in 2000).
12	Monitoring data from an operational incinerator in the UK recorded in 2001 (EA, 2001i).
13	Environment Agency spot sampling data from an operational incinerator in the UK recorded in 2001.
14	Continuous annual monitoring data carried out for the operator for the same plant as Reference 13 recorded in 2001.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction	Introduction Scope Sources Materials Recovery Facilities (ii		Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

Reference No.	Reference source
15	Continuous annual monitoring data carried out for the operator at an operational incinerator in the UK.
16	Review paper on the emissions from waste management options in the US. There is no comment on the derivation methodologies for data. Data from this source has not been used when calculating the preferred emission values, because of uncertainty over its accuracy, and because it refers to US plant. (Anon, 1996)
17	Review paper on incineration in the UK. Data on carbon dioxide emissions is calculated on the basis that the carbon content of MSW is 24% w/w. The carbon dioxide data from this report has not been used in the calculation of the preferred emission value.
18	Isle of Wight Incinerator – PI 2000.
19	Fabrellas B. <i>et al.</i> (1999) The Spanish Dioxin inventory: Proposal and preliminary results from municipal waste incinerator emissions. Organohalogen compounds 41
20	Marb C. <i>et al.</i> (1997) PCDD and PCDF in bottom ash from municipal solid waste incinerators in Bavaria, Germany. Organohalogen compounds 32
21	Energy from Waste Foundation (1999) Incinerator bottom ash: a review WRc, Harlow
22	Environment Agency (2002g) Solid Residues from Municipal Waste Incinerators in England and Wales

(81)

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Auction Scope Sources Materials Recovery Cin Facilities (in		Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

2.11 Small Scale Incineration of Pre-sorted Wastes with Energy Recovery

2.11.1 Introduction

Small scale incineration with pre-sorting of waste takes place at one site in the UK but has been suggested as having significant potential in future integrated waste management solutions. There are further plant under construction and others planned. Pre-sorting of waste takes place on-site, and emissions from this stage are represented in the MRF section of this chapter.

2.11.2 Sources of data; limitations; assumptions; areas of uncertainty

Information has been taken from the PI data for 2000 and monitoring data supplied by the operator for the same site. The errors, and uncertainty are as per those for the mass incineration data. Data pedigree is reduced because the data are taken from a single unit which may not be representative of other operations. The likely scale of emissions during abnormal operating conditions is similar to that described for large-scale incineration.

Introduction	Introduction		Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Introduction Scope Sources Recovery C Facilities (ii		Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary	

2.11.3 Emissions to air per tonne of waste

Substances	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Estimated Total UK Releases from Waste Processed during 2000 (T)	Comments	Data Pedigree
Nitrogen Oxides	1,269	1,587	1,983	24		Moderate (7)
Total Particulates	6	8	10	0.1		Moderate (7)
Sulphur Dioxide	16	20	25	0.3		Moderate (7)
Hydrogen Chloride	59	74	93	1		Moderate (7)
Hydrogen Fluoride	0.8	1	1.3	0.015		Moderate (7)
Volatile Organic Compounds	26	33	41	0.5		Moderate (7)
1,1 – Dichloroethane					No data	
Chloroethane					No data	
Chloroethene					No data	
Chlorobenzene					No data	
Tetrachloroethene					No data	
Methane					No data	
Cadmium	0.006	0.007	0.009	0.0001		Moderate (7)
Nickel	0.27	0.33	0.42	0.005	BRT ¹	Moderate (7)
Arsenic	0.027	0.033	0.042	0.0005	BRT	Moderate (7)
Mercury	0.017	0.021	0.027	0.0003		Moderate (7)
Dioxins and Furans (ng TEQ/tonne)	1,920	2,400	3,000	36		Moderate (7)
Dioxin-like polychlorinated Biphenyls (mg TEQ/tonne)					No data	
Carbon Dioxide (kg/tonne)	800	1000	1250	15		Moderate (7)

Table 2.30 Emissions to air from small-scale incineration

¹BRT: Below Reporting Threshold

Emissions to air from the small-scale incineration process under consideration differ from the values obtained for mass burn incineration in the previous section. For example, emissions of particulates and sulphur dioxide are lower from the small-scale unit, whereas emissions of VOCs, dioxins and furans and hydrogen chloride are lower from mass burn incinerators.

Based on a consideration of waste inputs, lower emissions of hydrogen chloride and dioxins and furans might have been expected from combustion of pre-sorted waste in the small-scale unit, because of the removal of plastics from the fuel. The higher emissions which are recorded in practice may reflect the abatement systems used at the small-scale facility. It should be noted that this in no way suggests that the small-scale facility exceeded the terms of its authorisation, or that there was any significant risk to health.



Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction Scope		Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Offset to emissions from electricity generation

Incineration is normally operated with energy recovery, resulting in the generation of electricity for export to the National Grid. Generating electricity in this way may be viewed as reducing the need to generate electricity from other sources, with a benefit in reductions in emissions from these sources. The estimated reduction in emissions as a result of avoided use of fuels is likely to be similar to that described above for mass burn incineration.

2.11.4 Emissions to land/groundwater per tonne of waste

Disposal of processed waste to landfill is shown in Table 2.31. Values are derived from PI data from 2000.

Table 2.31	Disposal of Residue from Small Scale incineration to Landfill
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Substance	Lower estimate (T residue / T waste processed)	Best estimate (T residue / T waste processed) 0.19	Upper estimate (T residue / T waste processed)	Estimated Total UK Releases from Waste Processed during 2000 (kT)	Data Pedigree
Non-special waste	0.15	0.19	0.24	2.8	Moderate (8)

2.11.5 Emissions to sewer/surface water per tonne of waste

There are no reported emissions to surface water. Emissions to sewer are shown in Table 2.32 below.

Substances	Lower estimate (g/T)	Best estimate (g/T)	Upper estimate (g/T)	Estimated Total UK Releases from Waste Processed during 2000 (T)	Comments	Data Pedigree
Cadmium	0.027	0.03	0.042	0.5	BRT	Moderate (8)
Mercury	0.003	0.004	0.0054	0.065	BRT	Moderate (8)

Table 2.32 Emissions to Sewer from Small-Scale Incineration

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.12 Landfill with Landfill Gas Flaring and/or Energy Recovery

2.12.1 Introduction

As set out in Chapter 1, the majority of MSW in England and Wales is disposed of to landfill.

The Landfill (England and Wales) Regulations 2002 came into force on 15 June 2002. These new regulations implement the Landfill Directive (Council Directive 1999/31/EC), which aims to prevent, or to reduce as far as possible, the negative environmental effects of landfill. The regulations will have a major impact on waste regulation and industry in the UK. The major impact of the Landfill Directive will be the change of composition and mass of MSW being landfilled. Currently about 60% of MSW is biodegradable, and landfill is therefore a major contributor to the production of the potent greenhouse gas methane, when landfilled. This will need to be progressively reduced over the period up to 2020.

This may have the effect of reducing the quantity, and changing the composition of the landfill gas produced, with similar changes to leachate production. It is difficult to predict what the exact nature of these changes will be on individual landfills: this is the subject of an Environment Agency research project into the composition of leachate from landfills in continental Europe. This section concentrates on current emissions from data taken over approximately the past 10 years. Even during this period the composition and quantity of waste entering landfills has varied.

Emissions from landfill processes differ from other processes considered, in terms of the timescale over which emissions arise following disposal to landfill. Emissions of landfill gas and leachate from biodegradable waste materials take place over a period of years following disposal. In contrast, emissions from other processes take place at the time the waste is processed. In this study, we have set out to evaluate the total emissions that will arise from waste landfilled at present. These emissions will take place over a period of years. By the same token, emissions from a landfill at any one time arise from waste deposited over the preceding years.

2.12.2 Emissions to air per tonne of waste

Landfill gas (LFG) is the principal component of emissions to air from landfill sites. It is an end product of the degradation of biodegradable wastes by principally anaerobic processes once the waste has been deposited to landfill. The composition of the gas varies according to the type of waste and the phase of degradation of the waste but typically it contains a large proportion of methane (typically 65%) and carbon dioxide (typically 35%) by volume. Small amounts (around 1% in total) of a range of trace components such as organic gases or vapours are also present, a number of which are potentially harmful to health.

There are a number of ways in which landfill gases and products of combustion are released to the atmosphere. These include:

 Fugitive gas emissions from passive venting to atmosphere through purpose built vents, cracks in the capping material, or through active and uncapped areas of the site. These emissions will not experience any pre-treatment before release. Fugitive emissions may also occur via diffusion through the landfill cap. In appropriate circumstances, oxidation of the methane may occur, initially



Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

to carbon dioxide and also to biomass. Methane oxidation is widespread but difficult to assess numerically with confidence.

- Collection using a gas extraction system and subsequent burning in flares to destroy flammable constituents and reduce environmental impacts in comparison to fugitive releases.
- Collection using a gas extraction system and utilised to provide heat or power using energy recovery plant which uses the landfill gas as a flammable fuel, therefore also destroying flammable components and reducing environmental impacts.

Emissions from a site will comprise a combination of these routes.

Although flaring and gas utilisation destroy flammable constituents, the destruction ciency is not 100% and hence consideration must be given to the environmental impacts of the release and also possible releases of the resulting combustion products.

The assessment of the emissions to air from landfill focuses on the release of landfill gas directly from the landfill, and combustion products and unburnt landfill gas released after flaring or energy utilisation. Any releases to air of substances associated with the plant machinery operated on site is not included in this assessment.

Methodology

This section gives a brief description of the methodology employed to calculate the mass release of substances from 1 tonne of MSW and any assumptions made throughout the assessment. The assessment has been split into three emissions components and are detailed below:

Landfill component 1: fugitive releases

The fugitive releases were assessed using the following equation:

$M = C \times G$

Where:

M = fugitive mass release of substances from waste in landfill (g of substance per tonne of waste)

C = concentration of substance in released landfill gas (g/m^3)

G = amount of landfill gas generated per tonne of landfilled waste $(m^3/tonne)$

No assessment of the emissions of landfill gas from operational areas of a landfill at the early stages of landfilling have been carried out, although the composition will be different from landfill gas generated in completed or restored areas. The significance of this is low as quantity of gas released from these areas during the first few months of landfilling is small compared to the quantity released during the active lifetime of the landfill.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

The rate of gas production varies considerably depending on the age of the waste and the conditions influencing degradation. Typically, in-situ gas generation rates for UK landfills are between 5 – 10 m³/tonne of waste per annum (Ref. 27). Therefore, emissions from waste that is landfilled are released over a much longer period than from other waste management facilities, often more than twenty years.

Landfill component 2: releases from flaring

The releases from the flaring of landfill gas were assessed using the following equation:

 $M_{f} = (C_{f} \times V_{f} \times G)$

Where

 M_f = Mass release of substances from flaring landfill gas (g of substance per tonne of waste),

 C_f = concentration in flare stack flue gas (g/m³),

 V_f = volume of flare stack flue gas per cubic metre of LFG combusted

(dimensionless),

G = amount of landfill gas generated per tonne of landfilled waste (m³/tonne),

Landfill component 3: releases from gas utilisation

The releases from gas utilisation using generating engines have been assessed using the following equation:

 $M_e = (C_e \times V_e \times G)$

Where

M = mass release of substances from landfill gas utilisation (kg of substance per tonne waste),

 C_e = concentration in engine exhaust gas (g/m³),

 V_e = volume of engine exhaust gas per cubic metre of LFG combusted

(dimensionless),

G = amount of landfill gas generated per tonne of landfilled waste (m³/tonne),

To carry out the assessment for Components 2 and 3, it was necessary to evaluate the bulk flow rate of flue gases resulting from the combustion of a given volume of landfill gas in a flare or engine. The methane concentration in landfill gas (C_{meth}) is typically in the range 45 to 60%, with values most commonly around 55%. The volume of flue gases generated from a given volume of landfill gas was estimated by summing the following items

Volume of unburnt landfill gas per m° of input gas:	(100% - Cmeth)
Volume of combustion products of methane per m ³ of inp	ut gas: 3 × Cmeth
Volume of nitrogen per m ³ of input gas:	7.57 × M
Volume of excess air per m ³ of input gas:	(1+ 7.57 C _{meth}) × (x / (20.9%-x))

where *x* is the proportion of oxygen in flue gases

Total volume of flue/exhaust gas (dry) per m³ of LFG = $20.9\% \times (1+7.57C_{meth}) / (20.9\%-X)$

It was possible to establish a likely range for V_e , the volume of flue gases resulting from a volume G of landfill gas, by considering the likely range of the two variable



Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

parameters (proportion of landfill gas which is methane, and proportion of oxygen in the flue gases). The likely volume of engine flue gases resulting from a volume G of landfill gas is 6.4 x G \pm 20%. The equivalent value for flare flue gases is V_f = 6.8 x G \pm 20%. Values for V_e and V_f are specified at 273K, dry gas.

Where possible the assessment has been carried out with emissions data corrected to the same conditions of temperature, pressure, oxygen and water content. The specific conditions are noted at the bottom of the result tables for each Component.

The three components above provide an estimate of the emissions generated by 1 tonne of municipal solid waste if the gas generated were wholly released to atmosphere via either fugitive releases, flaring or energy utilisation respectively. A further evaluation has been carried out to estimate the typical emissions from a landfill site where a combination of emissions pathways is more likely to represent actual conditions at landfill. The evaluation consists of two case studies and are described in more detail below:

- Case Study 1: It is assumed that 75% of the emissions from the waste are collected by a gas extraction system and combusted in a generating engine. The remaining 25% of emissions are released directly to atmosphere with no pre-treatment.
- Case Study 2: It is assumed that 75% of the emissions from the waste are collected by a gas extraction system and combusted by flaring. The remaining 25% of emissions are released straight to atmosphere fugitively with no pre-treatment.

Data sources

The data used for this assessment have been drawn from sources such as scientific research specifically in this area, emissions monitoring results from landfill sites in the UK and elsewhere and guidance produced by the Environment Agency or other bodies. Most of the data used relates to landfill sites within the UK.

The literature review focused on gathering information and data on the following parameters:

- The amount of landfill gas produced by waste from the degradation process;
- The typical composition of the landfill gas produced;
- The concentrations of substances released to atmosphere subsequent to combustion of the landfill gas in a flare or engine.

Data sources used are listed at the end of the section, together with the reference number ascribed to each source. The Environment Agency has recently funded the development of a model for carrying out site-specific estimates of landfill gas emissions, known as Gas-sim. In this study, we reviewed and incorporated where appropriate the source data which was used to develop the Gas-sim model, together with other published information.

Uncertainty

Variability and uncertainty have been dealt with in the same way as for incineration, except where stated otherwise. Uncertainties that will arise with particular reference to the assessment of emissions from landfill will occur in:

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of iological ch	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the tal ses of	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

- Uncertainty in the amount of landfill gas generated by 1 tonne of municipal solid waste. The value of 200 normalised cubic metres (Nm³) derived from References 6, 24 and 25 is based on measurements and calculations and is subject to uncertainty as the composition of MSW, particularly the biodegradable fraction, is not constant. Other factors, which may influence the amount of landfill gas generated, are the characteristics of the site and landfill design. For the purposes of this assessment the uncertainty is taken as the standard deviation of all the values reported for the generation of landfill gas. Where only an upper and lower range of values have been reported, the mean of the two extremes has been used as the estimate of the volume of gas. In this case the uncertainty was estimated as ± 40%.
- Uncertainty in the throughput of landfill gas and calculation of total volume of flue or exhaust gases. This is based on a theoretical calculation that relies on assumptions such as the methane, oxygen and water content of the input and flue gas based on information gathered during the review of data sources. An approximate error of ± 20% was determined by altering the values of methane, oxygen and water content using the range of values identified for each parameter from the review of data sources.
- Measurement uncertainty. Typically this could range from ± 10 to 50% depending on factors such as the methodology employed to carry out the measurements, the competence of the monitoring team and quality of analytical procedures used for the analysis. There are no established monitoring protocols for monitoring emissions from the surface of landfill, flares or engines (although draft methods have been published by the EA) and therefore the measured concentrations for any substance could have been recorded using different methods. Also, concentrations of substances from short-term or spot measurements may not be representative of long-term emissions as the composition of the landfill gas being combusted in the engine or flare is inherently variable. It has been assumed that the measurement uncertainty is ± 20% for all measurements.
- The variability of the data set of measurements for each substance. It is inappropriate to use the mean and standard deviation of the data set to assess the uncertainty within the data, because the data are typically not normally distributed. To overcome this, where there are 5 or more data points, the standard deviations of the logarithms of the measured values were used to obtain the estimated uncertainty. The overall uncertainty is the root-mean-square uncertainty from these three parameters.
- Where there are less than 5 data points, the best estimate has been based on the median of the data set. Determination of the range has been based on the combination of uncertainties associated with the data on a root-mean-square basis.

In addition, the emissions have been taken to represent emissions arising from a landfill containing material representative of municipal solid waste. In addition to influencing the volume of landfill gas produced, the actual nature of the waste will also influence the nature and quantity of substances present within the landfill gas. Most landfills will receive waste other than municipal solid waste (e.g. commercial and industrial waste. The emissions data presented in this Chapter is not exclusively derived from municipal solid waste.

The key data inputs used to evaluate data pedigree are the estimates of landfill gas generation from landfilled waste and the variability of emissions concentrations measurements.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.12.3 Emissions to air per tonne of waste

Estimates of the quantity of landfill gas generated by MSW are presented in Table 2.33. These estimates are used in deriving estimated emissions from fugitive releases of landfill gas (Table 2.34); flaring of landfill gas (Table 2.35) and power generation from landfill gas (Table 2.36).

Substance	Emissions (Nm³/T waste)	Data References	Comments
Landfill Gas	150	6	Landfill gas from biodegradeable fraction in MSW
	372	6	Theoretical calculation
	229	6	Theoretical calculation
	270	6	Calculated from Italian data
	120 – 160	6	Laboratory scale experiments
	190 – 240	6	Measured at landfills
	60 – 180	6	Measured at landfills
	222	6	Mean UK landfill yield
	135	6	Estimated average
	200	6	Estimated average
	100 – 200	6	Estimated average
	208	24	Estimated average
	200 – 250	25	Experimental studies
Value used in assessment	200 ± 74 (37%)		Approximate mean of all values and ranges reported above Data pedigree: Moderate (6)

 Table 2.33
 Determination of the amount of landfill gas generated per tonne of MSW

Note: The value of 200 m^3 gas per tonne of waste is a moderately reliable estimate of the average amount of gas produced per tonne of MSW across UK landfills as a whole. There will be significant variations in this value from site to site.

Introduction		Review on Emi	of Information ssions	Review Epidem Researc	of iological ch	Quantifica Conseque Emissions	tion of Health ences of	Quantification Environment Consequence Emissions	on of the tal ces of	Context for Qu Health and Environmental Review of Pub Perception Iss	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.34 Component 1 – emissions from fugitive releases

Substance	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Data Refs.	Comments	Data Pedigree
Nitrogen oxides					Not emitted	
Total Particulates					No data	
Sulphur dioxide					Not emitted	
Hydrogen chloride	0.03	0.2	1.1	18, 20	Weighted average of measurements from both reference sources	Moderate (6)
Hydrogen fluoride	0.005	0.04	0.3	20	Weighted average of measurements from both reference sources	Moderate (6)
Total VOCs	12	25	55	18	Only 1 measurement	Moderate (6)
1,1-dichloroethane	0.13	2.7	53	18, 26	*Estimate based on Ref 26 value	Moderate (6)
Chloroethane	0.05	1.0	21	18, 26	*Estimate based on Ref 26 value	Moderate (6)
Chloroethene	0.06	1.1	22	18, 26	*Estimate based on Ref 26 value	Moderate (6)
Chlorobenzene	0.12	2.4	48	18, 26	*Estimate based on Ref 26 value	Moderate (6)
Tetrachloroethene	0.17	3.3	67	17, 18, 20, 26	*Estimate based on Ref 26 value	Moderate (6)
Methane	50000	75000	120000	1, 6, 9, 10, 17, 18	Average of a range of values	Moderate (6)
Cadmium					No data	
Nickel					No data	
Arsenic					No data	
Mercury					No data	
Dioxins and furans					No data	
Poly-chlorinated biphenyls					No data	
Carbon dioxide	80000	130000	210000	1, 6, 9, 10, 17, 18		Moderate (6)
Benzene	0.012	0.24	4.8	26	*Estimate based on Ref 26 value	Moderate (6)

Note: assessment based on emission concentrations at 273K, 101.3 kPa. These emissions will take place over a number of years. *Data Reference 26 is based on data from 81 sites and 45,000 records, for pollutants reported in Ref 26 the best estimate is based on the Ref 26 value. The basis for selection of substances in this table is set out in Section 2.2

Introduction		Review on Emi	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.35 Component 2 - emissions from landfill gas flaring

Substance	Lower estimate (g/T except	Best estimate (g/T except	Upper estimate (g/T except	Data Refs.	Comments	Data Pedigree
	stated otherwise)	stated otherwise)	stated otherwise)			
Nitrogen oxides	54	100	200	18, 27	Average of a range of values	Moderate (6)
Total Particulates	2	8	18	18	Average of 3 measurements	Moderate (6)
SO _x (as SO ₂)	47	120	330	18, 27	Average of a range of values	Moderate (6)
Hydrogen chloride	6	19	54	18, 27	Average of a range of values	Moderate (6)
Hydrogen fluoride	1	4	12	18, 27	Average of a range of values	Moderate (6)
Total VOCs	0.4	1.7	7.2	18, 27	Average of a range of values	Moderate (6)
Non-methane VOCs	0.4	1.9	11	18, 31	Average of 6 measurements	Moderate (6)
1,1- dichloroethane					No data	
Chloroethane					No data	
Chloroethene					No data	
Chlorobenzene					No data	
Tetrachloroethe ne	0.0001	0.008	0.04	18	Median of one range reported in reference	Moderate (6)
Methane	0	400	6000	10, 18	Average of 3 measurements	Moderate (6)
Cadmium					No data	
Nickel					No data	
Arsenic					No data	
Mercury					No data	
Dioxins and furans	4 ng TEQ/T	74 ng TEQ/T	1,500 ng TEQ/T	18, 31	*Estimate based on the median of the two Ref. 31 values	Moderate (6)
Poly-chlorinated biphenyls					No data	
Carbon dioxide	120000	220000	400000	10, 18, 27	Average of a range of values	Moderate (6)

Note: assessment based on emission concentrations at 273K, 101.3 kPa, 3% O₂, dry. These emissions will take place over a number of years. * The best estimate has been based on data from Ref. 31.

Introduction	Introduction		Review of Information on Emissions		Review of Qua Epidemiological Con Research Emi		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.36 Component 3 - emissions from landfill gas generating engine

Substance	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated	Data Refs.	Comments	Data Pedigree
Nitrogen oxides	400	900	2100	1, 6, 10,		Moderate (6)
	+00		2100	20	Average of large	
Total Particulates	2	7	23	1, 6, 10, 20	number of measurements at	Moderate (6)
SO _x (as SO ₂)	30	70	170	1, 6, 10, 20	several sites	Moderate (6)
Hydrogen chloride	0.9	4	22	1, 6, 10	Average value from a range of measurements	Moderate (6)
Hydrogen fluoride	0.9	4	20	1	Average of 7 measurements	Moderate (6)
Total VOCs					No data	
Non-methane VOCs	6	30	160	1	Average of 10 measurements	Moderate (6)
1,1-dichloroethane					No data	
Chloroethane					No data	
Chloroethene					No data	
Chlorobenzene					No data	
Tetrachloroethene	0.09	0.2	0.4	10	Average of measurements at 3 sites	Moderate (6)
Methane	1,000	2,000	4,000	1	Average of 7 measurements	Moderate (6)
Cadmium	0.02	0.1	0.60	20		Moderate (6)
Nickel	0.007	0.013	0.02	20	Average of	Moderate (6)
Arsenic	0.0008	0.0016	0.003	20	up to 8 sites	Moderate (6)
Mercury	0.0008	0.0016	0.003	20		Moderate (6)
Dioxins and furans	14 ng TEQ/T	190 ng TEQ/T	2,500 ng TEQ/T	1, 6, 10, 20	Average of large number of measurements at several sites	Moderate (6)
Poly-chlorinated biphenyls					No data	
Carbon dioxide	190000	350000	650000	1, 10	Average value from a range of measurements	Moderate (6)

Note: assessment based on emission concentrations at 273K, 101.3 kPa, 3% O₂, dry. These emissions will take place over a number of years.

Introduction		Review on Emi	of Information ssions	Review Epidem Researc	Review of Epidemiological Research		Quantification of Health Consequences of Emissions Quantification of the Environmental Consequences of Emissions Mechanical Apparchic Confection Indiport		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions		
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Substance	Case Study 1: 75% engines and 25% fugitive			75% fla	Case Study 2 ring and 25%	fugitive	Data
	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Lower estimate (g/T except where stated otherwise)	Best estimate (g/T except where stated otherwise)	Upper estimate (g/T except where stated otherwise)	Pedigree
Nitrogen oxides	300	680	1,580	41	75	150	Moderate (6)
Total Particulate Matter	1.5	5.3	17	1.9	6.1	13	Moderate (6)
Sulphur oxides	23	53	130	35	90	260	Moderate (6)
Hydrogen chloride	0.68	3.0	17	4.8	14	41	Moderate (6)
Hydrogen fluoride	0.68	3.0	15	0.77	2.7	9.2	Moderate (6)
Total VOCs	2.9	6.4	14	3.3	7.6	19	Moderate (6)
NMVOCs	4.5	23	120	0.30	1.4	8.1	Moderate (6)
1,1-dichloroethane	0.033	0.66	13	0.033	0.66	13	Moderate (6)
Chloroethane	0.013	0.26	5.2	0.013	0.26	5.2	Moderate (6)
Chloroethene	0.014	0.28	5.6	0.014	0.28	5.6	Moderate (6)
Chlorobenzene	0.030	0.59	12	0.030	0.59	12	Moderate (6)
Tetrachloroethene	0.11	0.98	17	0.042	0.84	17	Moderate (6)
Methane	13000	20000	33000	13000	19000	35000	Moderate (6)
Cadmium	0.011	0.071	0.45	No data	No data	No data	Moderate (6)
Nickel	0.0056	0.0095	0.016	No data	No data	No data	Moderate (6)
Arsenic	0.00058	0.0012	0.0025	No data	No data	No data	Moderate (6)
Mercury	0.00060	0.0012	0.0024	No data	No data	No data	Moderate (6)
Dioxins and furans	11 ng TEQ/T	140 ng TEQ/T	1,900 ng TEQ/T	2.8 ng TEQ/T	55 ng TEQ/T	1,100 ng TEQ/T	Moderate (6)
Polychlorinated biphenyls	No data	No data	No data	No data	No data	No data	
Carbon dioxide	163000	300000	540000	110000	200000	350000	Moderate (6)
Benzene	0.000003	0.000060	0.0012	0.000003	0.000060	0.0012	Moderate (6)

Table 2.37 Summary of results for case studies

Note: These emissions will take place over a number of years.

Conclusions

The estimated mass of substance released to air per tonne of MSW landfilled is summarised in Tables 2.37 and 2.38.

The range and degree of uncertainty reported for each substance highlights the difficulties in trying to ascertain a single value to represent the release of a substance resulting from landfilling a specific amount of waste. The variability of several parameters, such as the exact composition of the waste, measurement and chemical analysis techniques, the type of flare or generating engine used to control the releases and whether the equipment was operating correctly or efficiently during measurements leads to high uncertainty ranges as can be seen in Tables 2.34 to 2.36.



Introduction		Review on Emi	of Information ssions	Review Epidem Researc	Review of Epidemiological Research Mechanical According		tion of Health inces of	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

There were no data available for biohazards or PCBs. The amount of data for other substances such as metals and VOCs was also limited in some cases.

Substance	Best estimate (g/tonne of waste)		
	Component 1 - Fugitive releases	Component 2 - Flaring	Component 3 – Engine
Nitrogen oxides	Not emitted	100	900
Total Particulate Matter	No data	8	No data
Sulphur oxides	Not emitted	120	70
Hydrogen chloride	0.2	19	4
Hydrogen fluoride	0.04	4	4
Total VOCs	25	1.7	No data
NMVOC	No data	1.9	30
1,1-dichloroethane	2.7	No data	No data
Chloroethane	1.0	No data	No data
Chloroethene	1.1	No data	No data
Chlorobenzene	2.4	No data	No data
Tetrachloroethene	3.3	0.008	0.2
Methane	75,000	400	2,000
Cadmium	Likely to be similar to engine	Likely to be similar to engine	0.1
Nickel	Likely to be similar to engine	Likely to be similar to engine	0.013
Arsenic	Likely to be similar to engine	Likely to be similar to engine	0.0016
Mercury	Likely to be similar to engine	Likely to be similar to engine	0.0016
Dioxins and furans	No data	74 ng TEQ/T	190 ng TEQ/T
Polychlorinated biphenyls	No data	No data	No data
Carbon dioxide	130,000	220,000	350,000
Benzene	0.24	No data	No data

 Table 2.38
 Emissions of substances per tonne of MSW for each emission component

Note: these emissions will take place over a number of years.

Estimated total UK releases from waste processed in 2000

Total releases from waste processed in the UK during 2000 have been estimated based on a total of 27.1 MT of MSW landfilled in 2000/01. This total is based on 22.1 MT in England (DEFRA websites, 21 August 2002), 2.5 MT in Scotland (SEPA, Waste Data Digest 2002), 1.5 MT in Wales (DEFRA 2003c, Digest of Environmental Statistics) and 1.0 MT in Northern Ireland (based on the 1998/99 data from the Department of the Environment Northern Ireland, Waste Management Strategy for Northern Ireland, 2002). The estimated total releases represent the total quantity of a pollutant released from the waste landfilled in 2000 during decomposition over the coming years. The proportion of landfill gas released fugitively, via flaring and via utilisation is based on the proportions of gases released by each route during 2002 in the LQM 2003 report (Data Ref. 24). The proportion of landfill gas that is combusted, using flares and engines, during 2002 has been interpolated from data in the report as 67%, leaving the remainder to escape fugitively.

Introduction		Review on Emi	of Information ssions	mation Review of Quantific Epidemiological Consequ Research Emission		Quantifica Conseque Emissions	tion of Health inces of	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

It is probable that the proportions of landfill gas that are released from each emission route will change due to drivers such as the Landfill Directive. It is anticipated that improvements in practice and technology will lead to improved capture rates of landfill gas reducing fugitive emissions, and a drive towards utilisation of landfill gas to generate electricity will cause the operational capacity of engines to increase more quickly than flaring capacity. As such it is believed that the estimated total releases from waste processed during 2000 are an over estimate of fugitive emissions and under estimate the emissions from utilisation and flaring of LFG.

Substances		Component		Total	Data
	Fugitive (T)	Flaring (T)	Engines (T)	(T)	Pedigree
Nitrogen Oxides	Not emitted	1,300	4,800	6,100	Moderate (6)
Total Particulates	No data	100	No data	100	Moderate (6)
Sulphur Dioxide	Not emitted	1,500	380	1,900	Moderate (6)
Hydrogen Chloride	1.8	240	21	260	Moderate (6)
Hydrogen Fluoride	0.32	45	21	66	Moderate (6)
Volatile Organic Compounds	230	22	No data	250	Moderate (6)
Non-methane VOCs	No data	40	160	200	Moderate (6)
1,1 - Dichloroethane	24	No data	No data	24	Moderate (6)
Chloroethane	9.4	No data	No data	9.4	Moderate (6)
Chloroethene	10	No data	No data	10	Moderate (6)
Chlorobenzene	21	No data	No data	21	Moderate (6)
Tetrachloroethene	30	0.11	1.1	31	Moderate (6)
Methane	680,000	5,100	10,700	700,000	Moderate (6)
Cadmium	No data	No data	0.51	0.51	Moderate (6)
Nickel	No data	No data	0.068	0.068	Moderate (6)
Arsenic	No data	No data	0.0086	0.0086	Moderate (6)
Mercury	No data	No data	0.0086	0.0086	Moderate (6)
Dioxins and Furans (TEQ)	No data	1.01 x 10 ⁻⁶	8.6 x 10 ⁻⁷	1.9 x 10 ⁻⁶	Moderate (6)
Polychlorinated Biphenyls	No data	No data	No data	No data	
Carbon Dioxide	1,200,000	2,800,000	1,900,00	5,900,000	Moderate (6)
Benzene	2.2	No data	No data	2	Moderate (6)

Table 2.39	Total UK	emissions	from	waste	processed	durina	2000
10010 2.00		01113310113		waste	processeu	uuring	2000

Note: These emissions will take place over a number of years.

Offset to emissions from electricity generation

Landfill is often operated with energy generation from combustion of landfill gas, resulting in the generation of electricity for export to the National Grid. Generating electricity in this way may be viewed as reducing the need to generate electricity from other sources, with a benefit in reductions in emissions from these sources. The estimated reduction in emissions as a result of avoided use of fuels is set out in Appendix 3. This suggests that there is a net reduction in emissions to air of

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

sulphur dioxide, particulates and benzene from the generation of electricity by the use of landfill gas, compared to emissions from UK power generation in 2001. There is a net increase in other emissions from the generation of electricity from landfill gas, compared to power generation in 2001.

2.12.4 Emissions to air during non-standard operating conditions

The most important factor which could affect emissions to air during non-standard operating conditions is the possibility of a breakdown in landfill gas combustion plant. Many of the larger UK landfill sites accepting MSW operate a system with landfill gas engines taking most or all of the collected gas, with a flare dealing with any remaining gas and/or operating in the event of a breakdown of the gas combustion plant. Because landfill gas is a complex mix of substances, it can be difficult to handle, resulting in occasional interruptions to the operation of combustion plant. The Environment Agency (2004) indicated that interruptions to the operation of landfill gas generating engines.

Under these circumstances, emissions to air would be represented by the data for "Component 1, Fugitive Release". Again, any short-term increase in emissions is not likely to have a significant effect on emissions averaged over a longer period such as a year. Landfills accepting MSW are regulated by the Environment Agency, with the aim of minimising the occurrence of such incidents.

It is now common practice to install landfill gas collection systems at landfill sites within six months of commencement of disposal in a particular cell. This does not occur at all sites, particularly those which have been licensed for some time. This would represent operation below current best practice, rather than operation under abnormal or non-compliant conditions. Again, this would result in an increase in the proportion of landfill gas emitted without combustion.

A complementary part of the control of landfill gas is the use of daily cover on active areas of a site and capping of completed parts of a site. If the materials used for daily cover are inappropriate, or are spread too thinly, this can result in release of odours and other components of landfill gas from a site. Because the materials in a landfill are continually decomposing and reducing in volume, a cap may develop fissures or holes over a period of time. Again, this can result in releases of landfill gas. The increased use of surface emissions surveys will reduce the influence of these emissions, as fissures in the landfill surface will be found and repaired more quickly.

Gas within the body of a landfill could potentially catch fire. This could result in the release of combustion products of landfill gas and the waste itself. This is less likely to occur at sites with properly designed and operated landfill gas collection systems Fires occurring at landfill sites in the UK have been associated with disposal of tyres. Landfill fires can be best dealt with by loading the surface of the landfill with earth to compress the waste and remove access to oxygen.

Problems with landfill odours have occurred as a result of disposal of inappropriate wastes at landfill sites. The most notable example of this was the disposal of plasterboard and other wastes containing relatively high levels of sulphur. The reducing conditions within landfills where this occurred resulted in the generation and release of hydrogen sulphide in landfill gas, with consequent odour problems. Disposal of liquids at landfill sites is no longer permitted, which has reduced the potential for odours to arise from liquid disposal.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.12.5 Data sources for releases to air from landfill

Ref	Author	Title
1	Gillett, AG, Gregory, RG, Blowes, JH and Hartnell G.	Land Quality Management Ltd report No. 158 to EB Nationwide, 2002.
2	B Croft, R Smith, M Caine, et al	The Brogborough Test Cells: Conclusions from a 14- year field scale landfill gas experiment, 2001
3	Borjesson and Svensson	Seasonal and diurnal methane emissions from a landfill and their regulation by methane oxidation, Waste Management Research (1997) 15, 33 – 54
4	JT Houghton, BA Calander and SK Varney	Climate Change. The supplementary Report to the IPCC Scientific Assessment, CUP, 1992
5	J Heerenklage and R Stegman	Comparison of test systems for determination of the gas potential from waste, 2001
6	Procter & Gamble, PR White, M Frankle P Hindle	Integrated Solid Waste Management, A life Cycle Inventory
7	AEA Technology (Simon Thresh, Anton van Santen)	Implications of the EC Landfill Directive and the draft waste strategy on UK greenhouse gas emissions: Preliminary Study. AEAT/ED21195/Final report, 2000 (Confidential report)
8	D Barry, N Ala et al, RH Gregory, C Harries	Minimising Methane Emissions from Municpal Landfills, Technical Forum 5 th April 2001, Bedfordshire
9	Department of Environment (M Fleet, CP Young et al)	The Technical Aspects of Controlled Waste Management: Landfill monitoring at the Ingham landfill sites 1974-1992
10	Environment Agency, Research and Development	Life Cycle Inventory Development for Waste Management Operations: Landfill P1/392/3, 2000d
11	USEPA, Office of Solid Waste and Emergency Response	Greenhouse Gas Emissions From Management of Selected Materials in Municipal Solid Waste, 1998
12	R. Pipatti and I Savolainen	Role of Energy Production in the control of greenhouse gas emissions from waste management, Energy Conversion and Management, vol. 37 no 6 pp 1105-1110
13	S Teneggi, V Nizzoli, E Bertolini	Monitoring of air emissions from a landfill capping and evaluation of effects on the biogas collection system efficiency, from 8 th International Waste Management and Landfill Symposium, Vol. 1, 2001
14	C R Harries, CJ Cross, R Smith	Application of Biochemical Methane Potential (BMP) Testing to the study of MSW decomposition in a municipal soild waste lysimeter, from 8 th International Waste Management and Landfill Symposium, Vol. 1, 2001
15	HJ Ehrig, I Krumpelbeck	The emission behaviour of old landfills in the aftercare phase, from 8 th International Waste Management and Landfill Symposium, Vol. 1, 2001
16	L Luning, A.A Boerboom et al	Evaluation of Effectiveness of Methane Emission Reduction, from 8 th International Waste Management and Landfill Symposium, Vol. 1, 2001
17	Environment Agency, Research and Development	Development of an Odour Attenuation System for Landfill Gas Emissions, CWM 100/93, 1993
18	Environment Agency, Research and Development	Guidance on the emissions from different types of landfill gas flares, CWM 142/96A, 1997b



Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Ref	Author	Title
19	Environment Agency, Research and Development	A framework to assess the risks to human health and the environment from landfill gas, Technical report P271, 1999b
20	Environment Agency	Measurement data from 8 landfill sites operating engines CONFIDENTAL DATA (2003k)
21	Environment Agency/WS Atkins	Methane emissions from different Landfill categories, March, 1997c
22	AEA Technology	Methane emissions from UK Landfills, March 1999
23	Environment Agency, Research and Development	The evaluation of possible health risks to landfill site workers from exposure to Gaseous waste emissions (Landfill Gas), P257, 2000e
24	DEFRA/LQM	Methane Emissions from Landfill Sites in the UK, LQM Report No. 443/1, January 2003b
25	Environment Agency, Research and Development	Appraisal of Hazards from Gas Producing Landfills CWM016/90, 1990
26	Environment Agency, Research and Development	Investigation of the Composition and Emissions of Trace Components in Landfill Gas, 2002e
27	Environment Agency, Research and Development	Guidance on the management of landfill gas (Draft for consultation), November 2002d
28	SEPA	Waste data digest, 2002
29	DEFRA	Digest of Environmental Statistics, February 2003
30	Department of the Environment Northern Ireland	Waste Management Strategy for Northern Ireland, 2000
31	EA, Research and Development	<i>Guidance for monitoring enclosed landfill gas flares</i> (Draft for consultation) December 2002a

2.12.6 Landfill emissions to sewer, groundwater and surface water

Introduction

Emissions from a landfill to sewer, groundwater and surface water arise from the production of leachate. Technology to control landfill leachate has advanced significantly during the past 25 years. The regulatory requirements have also changed and now require that this technology is incorporated into landfill construction to minimise emissions and harm to human health and the environment. Groundwater and surface water are environmental receptors but may also be pathways to a human receptor.

The Landfill Directive (1999/31/EC) implemented by the Landfill Regulations (England and Wales) (Statutory Instrument 2002 No. 1559) requires that all landfills receiving non-hazardous waste (which includes MSW) must have a barrier system to contain leachate. This must comprise:

- a drainage blanket to allow collection of leachate;
- a leachate sealing system including an artificial liner;
- a geological barrier comprising a low permeability mineral layer, extending under the base and up the sides of the landfill.

Introduction		Review on Emis	Review of Information on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

No barrier system can prevent all leachate passing through it. A risk assessment is frequently carried out for landfills accepting non-hazardous waste (e.g. MSW), to ensure compliance with the Groundwater Directive (80/68/EEC) and the Water Framework Directive (2000/60/EC). The Landfill Directive also requires that water from precipitation (rainfall) is prevented from entering the landfill and that surface water and/or groundwater are prevented from entering the landfilled waste. These requirements necessitate the construction of a cap following completion of landfilling operations in addition to barriers under the base and sides as discussed above. This cap should comprise a sealing layer, surface water drainage system and cover soils (Environment Agency Landfill Directive Regulatory Guidance Note 6.0 (version 3.0 June 2002h)).

It is now common practice to reduce infiltration and leachate production to a minimum, and thereby the long term costs associated with leachate removal and treatment. This is achieved by the installation of a geomembrane within the cap, which severely limits the production of leachate once the site is capped. Leachate is therefore likely to be produced in significant quantities only during operational phases of landfilling before the permanent cap is installed or from older closed landfills with less fully engineered caps. Leachate production is also minimised by progressive capping during the operational phase and in some cases by recirculation of leachate to utilise the full absorptive capacity of wastes.

The introduction of these strict engineering barrier controls in the construction of modern landfills is the result of a progressive change in landfill design to minimise both the production and fugitive emission of leachate from landfills. Nevertheless there will be finite emissions from landfills because no barrier or sealing system can completely prevent the entry of water into the waste, and formation of leachate.

This leachate will be removed by extraction of leachate for treatment and discharge, either at an on-site treatment plant or at an off-site sewage treatment works (STW). The residual level of leachate on the liner system, as the leachate is drained to the removal points results in a small but finite seepage through the basal barrier and into the ground.

The rate of seepage is limited by design, management and inspection to a level which can be safely accommodated by the surrounding environment.

Approach to estimating landfill leachate emissions

The quantification of the emission of substances in leachate is based on the conceptual model illustrated below:
Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary





The majority of landfill leachate is discharged following treatment in an on-site process, and/or at an off-site sewage works. A small proportion of leachate is released via the landfill lining system to land or groundwater.

This study takes a three stage approach to quantify these emissions:

Stage 1 : Identify concentrations of relevant substances in landfill leachate

Stage 2 : Estimate the amount of leachate collected for treatment

Stage 3 : Estimate the amount of leachate which is discharged to land/groundwater

Stage 4 : Based on the concentration of components of interest in leachate, estimate the quantities of substances discharged following treatment, and the quantities discharged to land/groundwater. The mass of any substance emitted in leachate to treatment or to land/groundwater per tonne of waste is in principle calculated as follows:

Mass released (g/T) = Quantity of leachate released per tonne waste $(m^3/T) \times Concentration of substance in leachate (g/m³)$

It should be noted that substances seeping through the landfill liner will be attenuated (i.e. degraded to other substances) during transportation through soils and groundwater. The extent of attenuation varies from site to site, but the evidence from monitoring groundwater quality around landfills is that this substantially reduces the potential for emissions to groundwater to have adverse environmental effects.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Stage 1: Identification of concentrations of relevant substances in landfill leachate

The nature of landfill leachate is a function of waste types, solubility, state of decomposition and degradation. Rainfall input can serve to dilute and flush contaminants in addition to assisting in the degradation process by wetting the wastes. A wide range of substances may potentially be present in leachate necessitating selection of substances for assessment based on their potential for adverse effects on human health and the environment.

The choice of substances for quantification in landfill leachate has been based on research into the concentrations of substances on the Environment Agency's Pollution Inventory. These substances have been selected because they are recognised as of concern to human health or the environment or are an EPER (European Pollutant Emission Register) Substance for Water. A total of 77 substances appear in the most recent Pollution Inventory list (July 2002), for water.

The Environment Agency research reports used to identify substances in leachate are:

- Pollution Inventory discharges to sewer or surface waters from landfill leachates; Ref: REGCON 70, May 2001e;
- Updating the Landfill Leachate Pollution Inventory Tool; R&D Technical Report No. PI-496/TR(2), March 2003h.

These reports provide information on the quantities of Pollution Inventory substances in raw and treated leachates from a representative cross-section of UK landfill types including co-disposal, purely domestic and mixed domestic, commercial and non-hazardous industrial wastes. The reports identify that out of the 77 Pollution Inventory substances only 27 occur in more than 5% of leachate samples. These 27 substances have been selected as appropriate for evaluation as potentially significant in emissions of leachate from landfill sites.

The choice of substances and the concentrations used in this evaluation is based on the available data for leachates from UK landfills. This database includes analyses from up to 67 landfill sites for the 27 substances. The size of this sample and diverse nature and age of the wastes rules out any attempt to determine the average leachate composition during the leaching history of any particular waste. This view is supported by the statistical distributions for the 27 substances selected which all show a positive skew with mean concentrations significantly greater than the median. Table 2.40 below lists the mean and median concentrations for the 27 substances under consideration.

The median concentrations have been used as the best estimate of raw landfill leachate composition. The ratio of mean to median concentration has been used to provide an estimate of the likely uncertainty of these estimates.

The percentage removal of substances during treatment are given in Table 2.21. It has been assumed that treated leachate discharged to sewer will be subject to biological treatment and that leachate discharged to surface water will be subject to biological treatment followed by polishing. The use of these estimates introduces an additional element of uncertainty into the release estimates. This is considered likely to be an uncertainty of the order of $\pm 30\%$ in the estimated proportion remaining in the leachate, with the consequential effect as set out in the table.

The data do not permit any refinement of this study to investigate the effect of any initial rise and subsequent fall in concentration during the evolution of leachate with



Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health ences of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

age of the waste. No quantitative investigation has been carried out of the potential variations in leachate quality which may arise from the implementation of the Landfill Directive requirement for progressive reduction in Biodegradable Municipal Waste. This will result in a national reduction in emissions of leachate from MSW, but the implementation of the reductions through a tradable permitting scheme is likely to produce wide variations in types of landfill and wastes accepted as operators are likely to concentrate individual waste streams in dedicated sites.

Introduction		Review on Emi	of Information ssions	Review Epidem Researd	of iological ch	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.40: Composition of raw and treated landfill leachate

Selected substances (see text)		Raw lead	hate		Follo	owing treatment	and discharg	ge to sewer	Followin	ng treatment and	d discharge t	o surface water
	Median concentration	Mean Concentration	Uncertainty factor	Data Pedigree	% Removal	Median Concentration	Uncertainty factor	Data Pedigree	% Removal	Median Concentration	Uncertainty factor	Data Pedigree
Aniline (µg/L)	<1	<1.46	1.5	Good (9)	80	<0.20	2.0	Moderate (5)	90	<0.10	2.0	Moderate (5)
Methyl tertiary butyl ether (µg/L)	<1	<1.38	1.4	Good (9)	0	<1.00	1.9	Moderate (6)	0	<1.00	1.9	Moderate (6)
Chloride (mg/L)	1145	1425	1.2	Good (9)	0	1145	1.8	Moderate (6)	0	1145	1.8	Moderate (6)
Cyanide (as CN) (mg/L)	<0.05	<0.05	2	Good (9)	0	<0.05	2	Moderate (6)	0	<0.05	2	Moderate (6)
Di(2ethyl hexyl)phthalate (µg/L)	<1	4.25	4	Good (9)	90	<0.1	4	Moderate (5)	95	<0.05	4	Moderate (5)
Ethylbenzene (µg/L)	<10	19	2	Good (9)	80	<2	2	Moderate (5)	80	<2	2	Moderate (5)
Fluoride (mg/L)	0.65	0.86	1.3	Good (9)	0	0.65	1.9	Moderate (6)	0	0.65	1.9	Moderate (6)
Methyl chlorophenoxy acetic acid (μg/L)	<0.1	0.69	7	Good (9)	95	<0.005	7	Moderate (5)	95	<0.005	7	Moderate (5)
Dichloromethane (µg/L)	<1	42.8	43	Good (9)	NA	<1	43	Poor (4)	NA	<1	43	Poor (4)
Nitrogen (Total) (mg/L)	364	629	2	Good (9)	NA	364	2	Poor (4)	NA	364	2	Poor (4)
Organo-tin (μg/L)	0.2	0.3	2	Good (9)	0	0.2	2	Moderate (6)	0	0.2	2	Moderate (6)
Phenols (mg/L)	0.03	0.35	12	Good (9)	99	0.0003	12	Moderate (5)	99	0.0003	12	Moderate (5)
Phosphorus (mg/L)	3	3.9	1.3	Good (9)	2	2.94	1.8	Moderate (5)	2	2.94	1.8	Moderate (5)
Polycyclic aromatic hydrocarbons (μg/L)	<5.25	<5.60	1.1	Good (9)	50	<2.63	1.7	Moderate (5)	50	<2.63	1.7	Moderate (5)
Nonyl phenol (µg/L)	1	4.9	5	Good (9)	95	0.05	5	Moderate (5)	95	0.05	5	Moderate (5)
Biphenyl (µg/L)	0.1	0.46	5	Good (9)	60	0.04	5	Moderate (5)	95	0.005	5	Moderate (5)
Mecoprop (µg/L)	11	21.8	2	Good (9)	99	0.11	2	Moderate (5)	99.5	0.055	2	Moderate (5)
Naphthalene (µg/L)	0.46	3.04	7	Good (9)	95	0.023	7	Moderate (5)	95	0.023	7	Moderate (5)
Pentachlorophenol and compounds (μg/L)	<0.1	0.32	3	Good (9)	50	<0.05	3	Moderate (5)	60	<0.04	3	Moderate (5)
Toluene (µg/L)	21	87	4	Good (9)	80	4.2	4	Moderate (5)	80	4.2	4	Moderate (5)
Xylenes (μg/L)	35	59	2	Good (9)	60	14	2	Moderate (5)	75	8.75	2	Moderate (5)
Arsenic (µg/L)	8	16	2	Good (9)	70	2.4	2	Moderate (5)	70	2.4	2	Moderate (5)
Chromium (µg/L)	50	92	2	Good (9)	30	35	2	Moderate (5)	30	35	2	Moderate (5)
Copper (µg/L)	11	26	2	Good (9)	50	5.5	3	Moderate (5)	50	5.5	3	Moderate (5)
Lead (µg/L)	<50	60	1.2	Good (9)	0	<50	1.8	Moderate (6)	0	<50	1.8	Moderate (6)
Nickel (µg/L)	60	159	3	Good (9)	20	48	3	Moderate (5)	20	48	3	Moderate (5)
Zinc (ua/L)	135	1246	9	Good (9)	70	40.5	9	Moderate (5)	70	40.5	9	Moderate (5)

The uncertainty factor gives an indication of the likely range in estimated concentrations. The likely range is obtained by multiplying and dividing the concentration value by the uncertainty factor. NA: data not available



Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & ic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Stage 2: Estimation of amount of leachate collected for treatment

As discussed above, the potentially significant emission routes for landfill leachate comprise abstraction of leachate from the landfill and seepage through the base and sides.

There are no published data giving actual quantities of leachate abstracted from landfills or seeping into the surrounding ground and underlying groundwater. Although the Pollution Inventory returns provide some information on quantities of leachate discharged to sewer, at the time of writing this is incomplete. There are no published data on volumetric discharges to groundwater and the Environment Agency has made no such assessment, on the basis that there is no significant impact nationally (communication from the Environment Agency to Enviros, 2003m).

The volume abstracted as raw leachate for treatment and/or discharge to sewer has therefore been derived independently, using two approaches to quantify the likely range of uncertainty. The first approach uses an estimate of the current annual quantity of leachate removed from landfills to obtain an indication of the total mass of each substance that can be expected from decomposition of the annual waste input to landfill. The second approach predicts the quantity of leachate produced during the life of "model" landfills assuming that they receive the total current annual MSW input and perform to currently expected standards of engineering.

First approach: In this approach, it is assumed that the current annual rate of leachate production from operational landfills is attributable to the current rate of disposal of wastes, of which a proportion is MSW. It is inherently assumed that the current types and nature of waste input are in a state of quasi equilibrium with current leachate quantity and quality. Post closure, leachate production is ignored because infiltration of water to the site will be small compared with that generated during the operational phase. Landfill sites are assumed to be progressively restored. Although leachate is generated in significant quantities from some closed sites at present, because of improvements in engineering design leachate generated from currently operational landfills after closure is likely to be significantly less than leachate currently being generated from older closed landfills.

The Environment Agency has requested Pollution Inventory returns from up to 1000 operational landfills, representing those that they consider are likely to be generating significant emissions of leachate.

The Pollution Inventory returns for emissions from landfills in 2002 are currently being received by the Environment Agency. At the time of undertaking this assessment, approximately one third of returns had been received by the Agency. When complete the information from these returns will provide:

- a database of actual emissions in raw and treated leachate from landfills (where these are above reporting thresholds);
- information on volumes of leachate discharged to sewer and surface waters where the Environment Agency estimator tool has been used.

This database may provide a valuable new source of information, but has not been available for use in the present study.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Current leachate production from landfills was estimated in a recent report (Enviros Aspinwall, September 2002) as 50 cubic metres per day of leachate from landfill sites of significant size. This rate of generation is typical of a large operational landfill but may tend to overestimate leachate production since it is likely to include phases capped with clay rather than the current practice of using a geomembrane.

Data in the Environment Agency Waste Statistics for 1998-99 indicate that at that time there were 1485 licensed landfills. More recent data for 2001 report that there were 622 landfill sites classified as co-disposal or taking household, commercial and/or industrial wastes although only a proportion of these receive significant quantities of biodegradeable wastes.

It is therefore estimated that the number of operational landfills likely to be producing significant quantities of leachate is likely to lie within the range of 350 to 950 sites with a best estimate of 650. In the absence of data to the contrary we have assumed that each site generates 50 cubic metres of leachate per day (this may be an overestimate, as noted above).

On this basis, it is estimated that 12 ± 5 million cubic metres per year of leachate is produced from landfills taking significant quantities of MSW and biodegradeable wastes. MSW represents approximately one third of waste delivered to these landfills, and leachate attributable to MSW has been assumed to be in the same proportion, a total of 4 ± 2 million cubic metres per year.

Out of the total estimated volume of leachate produced it has been assumed that 2% is treated on site and discharged direct to surface water, 30% is treated on site and discharged to sewer, and 68% is discharged direct to sewer (these percentages are approximations in the absence of any published information and based on Enviros' experience of design of on-site leachate treatment facilities). This provides the following estimated leachate volumes from landfill of 27 million tonnes of MSW per year:

On site treatment and discharge to surface water: $80,000 \pm 40,000 \text{ m}3/\text{year}$ $0.003 \pm 0.0013 \text{ m}^3/\text{T}$

On site treatment and discharge to sewer: 1,200,000 \pm 500,000 m3/year 0.04 \pm 0.02 m³/T

Discharge to sewer without on-site treatment: 2,700,000 \pm 1,200,000 m3/year 0.10 \pm 0.05 m³/T

There are no available data on the total quantity of leachate seeping into ground and groundwater from current landfill sites. An estimate of leakage rates could theoretically be made on the basis of assumed landfill areas and liner characteristics. However, these calculations have not been carried out because of the variabilities inherent between different landfill types. Instead, emissions to groundwater have been estimated on the basis of a model landfill using the LandSim model (see below).

Second approach: In this approach, the leachate produced during the operational and post closure phases from a landfill constructed to current engineering standards has been quantified by carrying out water balance calculations on a conceptual landfill.



Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

A representative model calculation has been carried out for a landfill of twenty years operational lifetime. Landfill cells are assumed to be constructed to current design specifications which minimise both the total quantity of leachate produced and seepage to groundwater though the barrier system. The quantities of emissions from a conceptual landfill have been estimated, making a range of assumptions on the effectiveness of barrier systems to minimise seepage to groundwater and infiltration during the operational phase and closed phases following capping and closure.

Leachate production will vary geographically across the country, being higher in the north and west than in the south and east. This variation reflects both the range of total rainfall at the site, together with variations in the moisture content, and therefore absorptive capacity, of the waste received. Once capped with a geomembrane, the rate of infiltration through the cap will be the limiting factor in generation of leachate.

The inputs to this calculation are as follows:

- Rate of infiltration through the cap: a theoretical value of 2 mm per year is commonly used for modern membrane caps, and is adopted in this study. The potential effects of a 10-fold increase have also been considered.
- Total annual waste input to landfill of 27.1 million tonnes per annum of MSW;
- Rainfall infiltration of between 600 and 1000mm, representing the typical range of values experienced in the UK;
- Absorptive capacity varying from 5% to 10% (available absorptive volume as a percentage of waste volume received)
- ♦ Waste density of 0.85 T/m³;
- Site operational life of 20 years;
- Average waste depth of 15 metres, giving a total combined area covered by conceptual landfills of approximately 213 hectares each year;

A water balance calculation indicates that the total leachate production from these conceptual landfills due to the landfilling of 27.1 million tonnes per annum of MSW during a twenty year life is estimated to vary between 3000 and 3700000 cubic metres. The best estimate value is 2 ± 2 million cubic metres produced from the annual total of waste landfilled, or 0.068 ± 0.034 cubic metres per tonne of waste. This is about half the estimate obtained using the first approach.

The median and maximum estimated values are increased by approximately 10% by a change in infiltration rate through the cap to 20 mm per year.

Summary: The estimated rates of leachate generated from MSW, derived from these two approaches are as follows:

- On site treatment and discharge to surface water: 60,000 ± 30,000 m3/year
 0.002 ± 0.001 m³/T
 Pedigree: Poor (4);
- On site treatment and discharge to sewer: 900,000 ± 400,000 m3/year
 0.033 ± 0.015 m³/T
 Pedigree: Poor (4);

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

- Discharge to sewer without on-site treatment: 2,000,000 ± 900,000 m3/year
 0.075 ± 0.03 m³/T
 Pedigree: Poor (4);
- Total volume of leachate:
 2,960,000 ± 1,300,000 m3/year
 0.110 ± 0.045 m³/T
 Pedigree: Poor (4).

Stage 3 : Estimate the amount of leachate which is discharged to land/groundwater

The quantity of leachate seeping into ground and groundwater from the model landfill can be estimated from the site area and expected performance of the barrier system. In a modern landfill site the quantity of leakage through the base can be reduced to less than 3mm per annum. This quantity amounts to 0.13 million cubic metres during the 20 year life of the conceptual landfill, or 0.005 cubic metres per tonne of waste, approximately 4% of the average leachate production during the operational phase.

Following the end of landfill operations, infiltration is reduced to approximately 2mm per annum provided that the cap retains integrity. This would be expected to seep through the base of the landfill. A post-closure period of 150 years has been used to evaluate leachate emissions, as leachate strength will reduce over this period, with the result that the majority of emissions are expected to take place over this timescale. As an upper case it has been assumed that all the leachate released is of a quality comparable to the operational phase and enters groundwater. Although it is possible that leachate production may subsequently increase, it has been assumed that this would be balanced by corresponding reductions in the substances under consideration due to their degradation although it is not possible to obtain field evidence to confirm this assumption. Also, no account has been taken of attenuation in the unsaturated zone which will serve to reduce the amounts of released substances reaching a point where they could potentially have adverse environmental effects.

Over this period, the total quantity of leachate released would be 0.64 million cubic metres. This represents around one third of the quantity produced during the operational phase.

Modern landfills are subject to risk assessments which require sites to be designed and operated such that there is no significant impact on groundwater. Consequently although over this long time period there is potential for leachate to seep out of the landfill, the effect of the substances under consideration should be insignificant if the risk has been accurately assessed during the landfill design phase and the site operated according to the requirements identified in the risk assessment.

Limitations and uncertainty

There are a number of limitations and considerable uncertainty in the above estimates. The primary limitation to accurate quantification of emissions from a unit quantity of waste is the large number of variables affecting leachate quality and quantity.

The following aspects of landfill setting, design and operation affect leachate quality:

 It is rare for a landfill to accept MSW alone: consequently the currently available leachate quality data is representative of a mixture of wastes which include non-MSW waste types;



Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

- current leachate quality is likely to reflect a different mix of wastes from that which will be permitted in a non-hazardous site under the Landfill Directive;
- leachate quality will improve following closure as the waste degrades and source term declines. However there is considerable uncertainty regarding the rate of change in quality within a relatively dry landfill
- Reduced infiltration will extend the period over which degradation products are released in leachate providing further opportunity for the substances under consideration to degrade.

The variables affecting estimates of leachate quantity include:

- infiltration into uncapped waste during the operational phase varies geographically;
- infiltration through the cap of a modern landfill is significantly less now than landfills constructed only 10 years ago. This may be typically 30 to 50mm/annum in older landfills and 2mm/annum in modern landfills with a geomembrane cap;
- reduced post-capping infiltration is likely to extend the period over which degradation products are released in leachate possibly to a period of many hundred years; these low rates of infiltration may be of the same order as those of seepage thought the base and therefore may account for all leachate arising from post-capping infiltration.

The key factors that have been taken into account within the assessment of pedigree are the assumptions used for leachate production and migration. The variability of monitoring data, and the factors applied to substance concentrations to represent the effects of treatment are also significant.

Stage 4 : Emissions to sewer and surface water

The quantities of selected Pollution Inventory substances arising from one tonne of MSW have been derived from the leachate volumes and trace substance concentration information set out in Stages 1 to 3 above.

Emissions to sewer will be treated on passage through the sewage treatment works. UK Water Industry Research Ltd has developed a protocol for estimating emissions of pollution inventory substances from sewage treatment works (UKWIR, 2001). Information in this report has been used to estimate the reduction in emissions likely to occur as a result of sewage treatment.

Emissions to sewer are set out in Table 2.41. Emissions to surface water and groundwater are set out in Table 2.42.

Introduction		Review on Emi	of Information ssions	Review Epidemi Researc	of iological ch	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

 Table 2.41:
 Estimated emissions from landfill to sewer

Selected substances (see text)	Estimated discharge from landfill to STW (g/T)	Data Pedigree	Estimated national discharge from landfill to STW (kg/year)	Uncertainty factor ¹	Estimated removal rate in STW	Estimated discharge rate from landfill via STW to receiving waters (g/T)	Data Pedigree	Estimated national discharge from landfill via STW to receiving waters (kg/year)
Aniline	<0.00008	Poor (4)	<2.2	3				
Methyl tertiary butyl ether	<0.00011	Poor (4)	<2.9	3				
Chloride	123	Poor (4)	3340585	3				
Cyanide (as CN)	< 0.005	Poor (4)	<146	3				
Di(2ethyl hexyl)phthalate	<0.00008	Poor (4)	<2.1	5				
Ethylbenzene	<0.0008	Poor (4)	<22	3				
Fluoride	0.07	Poor (4)	1896	3				
Methyl chlorophenoxy acetic acid	<0.00008	Poor (4)	<0.21	7				
Dichloromethane	<0.00011	Poor (4)	<2.9	43				
Nitrogen (Total)	39	Poor (4)	1061985	3				
Organo-tin	0.000022	Poor (4)	0.6	3	99.6%	8.6 × 10 ⁻⁸	Poor (3)	0.0023
Phenols	0.0023	Poor (4)	61	12				
Phosphorus	0.32	Poor (4)	8699	3				
Polycyclic aromatic hydrocarbons	<0.00048	Poor (4)	<13	2				
Nonyl phenol	0.00008	Poor (4)	2.1	5	99.999%	7.6 × 10 ⁻¹⁰	Poor (3)	0.000021
Biphenyl	0.000009	Poor (4)	0.24	5	94%	0.0000053	Poor (3)	0.014
Mecoprop	0.0008	Poor (4)	22	3	41%	0.00049	Poor (3)	13
Naphthalene	0.00004	Poor (4)	1.0	7	96%	0.0000014	Poor (3)	0.038
Pentachlorophenol and compounds	< 0.000009	Poor (4)	<0.25	4	86%	< 0.0000013	Poor (3)	<0.035
Toluene	0.0017	Poor (4)	46	5	98%	0.000034	Poor (3)	0.93
Xylenes	0.0031	Poor (4)	83	3	97%	0.00010	Poor (3)	2.7
Arsenic	0.0007	Poor (4)	18	3	11%	0.00060	Poor (3)	16
Chromium	0.005	Poor (4)	132	3	66%	0.0017	Poor (3)	45
Copper	0.0010	Poor (4)	27	3	67%	0.00034	Poor (3)	9.1
Lead	< 0.0054	Poor (4)	<146	3	51%	<0.0026	Poor (3)	<71
Nickel	0.006	Poor (4)	164	3	41%	0.0036	Poor (3)	98
Zinc	0.011	Poor (4)	309	9	62%	0.0043	Poor (3)	118

Note 1: The uncertainty factor gives an indication of the likely range in estimated emissions of trace components. The likely range is obtained by multiplying and dividing the estimated discharge by the uncertainty factor.



	Introduction		Review of Information on Emissions Materials Recovery		Review Epidemi Researc	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		n of the al es of	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
I	Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

 Table 2.42:
 Estimated emissions from landfill to surface water and groundwater

Selected substances (see text)		Discharge	e to surface water			Discharg	e to groundwater	
	Estimated discharge (g/T)	Data Pedigree	Estimated national emission from landfill (kg/year)	Uncertainty factor ¹	Estimated discharge (g/T)	Data Pedigree	Estimated national emission from landfill (kg/year)	Uncertainty factor ¹
Aniline	<0.0000022	Poor (3)	<0.006	3	<0.000024	Poor (3)	<0.06	4
Methyl tertiary butyl ether	<0.000022	Poor (3)	<0.06	3	<0.000024	Poor (3)	<0.6	4
Chloride	3	Poor (3)	68175	3	27	Poor (3)	730106	3
Cyanide (as CN)	<0.00011	Poor (3)	<3.0	3	<0.0012	Poor (3)	<32	4
Di(2ethyl hexyl)phthalate	< 0.0000011	Poor (3)	<0.0030	5	< 0.0000012	Poor (3)	<0.032	5
Ethylbenzene	< 0.000004	Poor (3)	<0.12	3	<0.000047	Poor (3)	<1.3	4
Fluoride	0.0014	Poor (3)	39	3	0.015	Poor (3)	414	4
Methyl chlorophenoxy acetic acid	< 0.00000011	Poor (3)	<0.00030	7	< 0.0000012	Poor (3)	< 0.0032	8
Dichloromethane	< 0.0000022	Poor (2)	<0.06	43	<0.000024	Poor (2)	<0.6	43
Nitrogen (Total)	0.8	Poor (2)	21673	3	8.6	Poor (2)	232104	4
Organo-tin	0.00000044	Poor (3)	0.012	3	0.000005	Poor (3)	0.13	4
Phenols	0.000007	Poor (3)	0.018	12	0.000007	Poor (3)	0.19	12
Phosphorus	0.006	Poor (3)	175	3	0.07	Poor (3)	1875	4
Polycyclic aromatic hydrocarbons	< 0.000006	Poor (3)	<0.16	3	<0.00006	Poor (3)	<1.7	3
Nonyl phenol	0.00000011	Poor (3)	0.0030	6	0.0000012	Poor (3)	0.032	6
Biphenyl	0.00000011	Poor (3)	0.00030	5	0.00000012	Poor (3)	0.0032	6
Месоргор	0.00000012	Poor (3)	0.0033	3	0.0000013	Poor (3)	0.035	4
Naphthalene	0.00000005	Poor (3)	0.0014	7	0.0000005	Poor (3)	0.015	7
Pentachlorophenol and compounds	0.0000009	Poor (3)	<0.0024	4	0.0000009	Poor (3)	<0.026	5
Toluene	0.000009	Poor (3)	0.25	5	0.00010	Poor (3)	2.7	5
Xylenes	0.000019	Poor (3)	0.52	3	0.00021	Poor (3)	5.6	4
Arsenic	0.0000053	Poor (3)	0.14	3	0.000056	Poor (3)	1.5	4
Chromium	0.00008	Poor (3)	2.1	3	0.00082	Poor (3)	22	4
Copper	0.000012	Poor (3)	0.33	3	0.00013	Poor (3)	3.5	4
Lead	<0.00011	Poor (3)	<3.0	3	<0.0012	Poor (3)	<32	3
Nickel	0.00011	Poor (3)	2.9	4	0.0011	Poor (3)	31	4
Zinc	0.00009	Poor (3)	2.4	10	0.0010	Poor (3)	26	10

Note 1: The uncertainty factor gives an indication of the likely range in estimated emissions of trace components. The likely range is obtained by multiplying and dividing the estimated discharge by the uncertainty factor.



Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

The majority of the leachate discharged to surface water and sewer arises from the operational phase in the second approach since it is assumed that post closure the small quantity of leachate produced will seep through the base and sides of the landfill. A reduction in this could be achieved by increasing the frequency of installing temporary capping and improving methods of recycling leachate. However this is already being optimised at most recently developed landfill sites. Trends which might increase leachate production are the potential for future reduction in the more absorptive wastes, e.g. paper and packaging. Loss of all absorptive capacity would increase the quantity of leachate produced by approximately 50% for the best estimate in the second approach.

In the estimated releases to land/groundwater, no account has been taken of attenuation in the unsaturated zone beneath the landfill. A modern landfill is subject to a risk assessment which takes attenuation into account, designed to ensure that there is no unacceptable impact on groundwater. Consequently although over the long post closure time period there is potential for a significant quantity of leachate to seep out of the landfill, the quantity of the substances under consideration will be acceptable in the local environment if the risk has been accurately assessed during the landfill design phase.

Emissions during non-standard operating conditions

The key issue to consider for leachate emissions during non-standard operating conditions would be a failure or fault in the lining system. The use of a liner system does not prevent emissions to groundwater, but does ensure that any releases take place at a very low rate. A leak in the lining system could conceivably result in an increased discharge of leachate to groundwater. The possibility and consequences of leakage are addressed as part of a site's groundwater risk assessment, such that if it were to occur, there is no significant risk of harm to human health or the environment. The head of leachate above the base of a landfill cell is normally required to be maintained below one metre to minimise the likelihood and consequences of any leakage of leachate. Boreholes are drilled at key locations around landfill sites to enable any such release to be detected at an early stage, ensuring that there is no significant impact locally. This means that, while landfill leachate or contaminated surface water is released from landfill sites from time to time, it is very rare for such releases to affect controlled surface waters.

Leachate stored or treated at a landfill site can become low in oxygen, resulting in the generation of odorous compounds such as sulphides. This can result in a significant odour from tanks used to store leachate. This is likely to contribute to odour complaints associated with landfill sites, as discussed in Appendix 2 and chapter 6.

An interruption to the operation of on-site treatment facilities is also possible. Under these circumstances, leachate would not be discharged directly to controlled waters, but would be taken off-site for treatment at a wastewater treatment works. As noted above, MSW landfills are regulated by the Environment Agency, with the aim of minimising the occurrence of such incidents.

Introduction		Review of Information on Emissions		Review Epidemi Researc	Review of Epidemiological Research		tion of Health nces of	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.13 Waste Transportation excluding Collection

Emissions from waste transportation were calculated on the basis of information in the National Atmospheric Emissions Inventory. This database contains emissions estimates for heavy goods vehicles in 2000. Information provided separately by the Department for Transport (DFT, 2002) indicates that mileage travelled by heavy goods vehicles involved in household waste management activities accounts for 0.49% of all vehicle movements.

On this basis, the following emissions associated with waste transportation have been estimated. The uncertainty associated with the NAEI data is given as between $\pm 7\%$ and $\pm 50\%$. A further uncertainty is associated with the proportion of vehicle mileage associated with waste management, estimated to be of the order of $\pm 20\%$.

With the increasing integration of waste and resource management, it is likely that transport will increase as wastes are multiply handled. This has not been addressed in this study.

Substance	Estimated annual UK emission (kg/year)	Estimated emission per tonne of waste processed (g/T)	Uncertainty factor	Data Pedigree
Nitrogen Oxides	1040000	31	1.2	Moderate (7)
Total Particulates	45000	1.3	1.5	Moderate (7)
Sulphur Oxides	3700	0.11	1.2	Moderate (7)
Volatile Organic Compounds	173000	5.1	1.2	Moderate (7)
Benzene	98	0.0029	1.4	Moderate (7)
Benzo(a)pyrene	0.9	0.000026	3.0	Moderate (7)
Dioxins and furans (Toxic equivalent)	0.0000013	3.8×10^{-11}	2.0	Moderate (7)
Carbon dioxide	39600000	1170	1.2	Moderate (7)
Carbon monoxide	311000	9	2.2	Moderate (7)

Table 2.43 Emissions from waste transportation

The uncertainty factor gives an indication of the likely range in estimated emissions. The likely range is obtained by multiplying and dividing the estimated emission by the uncertainty factor.

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health nces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

2.14 Summary

The following table sets out the areas where information has been collated in this chapter:

 Table 2.44
 Information collated on emissions from waste management

	Emissions to air	Solid residue	Sewer	Controlled waters/ groundwater	Re- use	To landfill	Leachable element of vitrified residue
MRF	No data	No data	None	None	No data	No data	None
Composting	No data	M/T; UKT	M/T; UKT	None	No data	No data	None
МВТ	M/T	M/T	M/T	None	No data	No data	None
Gasification/ pyrolysis	M/T	M/T	None	None	No data	M/T	M/T
Mass burn incineration	M/T; UKT	M/T; UKT	M/T; UKT	M/T	M/T; UKT	M/T; UKT	None
Incineration of pre- sorted waste	M/T; UKT	No data	M/T; UKT	None	No data	No data	None
Landfill	M/T; UKT	None	M/T; UKT	M/T; UKT	None	None	None
Transport	M/T; UKT	None	None	No data	None	None	None

M/T: Mass per tonne MSW estimates provided; UKT: UK Total estimates provided

The following tables provide a summary of estimated emissions from waste management facilities:

- A summary of emissions to air from waste management facilities;
- A summary of estimated solid residues arising from waste management facilities is given below. This may comprise emissions to land (e.g. compost); re-use (e.g. incinerator ash re-used in building blocks); or disposal in landfill (e.g. incinerator air pollution control residues);
- A summary of emissions to sewer from waste management facilities;
- A summary of emissions to surface water from waste management facilities.

Some of the waste management facilities covered in the following tables are capable of handling the entire waste stream – MRFs, mass burn incineration and landfill. In contrast, the other facilities can only handle a fraction of the waste stream. Hence, the waste treated at these facilities is likely to differ in composition, making a direct comparison across facilities less straightforward.

Introduction		Review of Information on Emissions Materials Sources Recovery		Review Epidem Researd	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

 Table 2.45
 Emissions to air from waste management facilities (grams per tonne of MSW)

Substance	Windrow composting	MBT	Anaerobic Digestion	Pyrolysis/ gasification	Mass burn Incineration	Small scale Incineration / pre-sorting	Landfill / engines	Landfill/flaring	Transportation
Nitrogen Oxides	Not likely to be emitted	72.3 M(5)	188 M(8)	780 M(8)	1600 G(9)	1587 M(7)	680 M(6)	75 M(6)	31 M(7)
Total Particulates	175 P(3)	No data	No data	12 M(8)	38 G(9)	8 M(7)	5.3 M(6)	6.1 M(6)	1.3 M(7)
Sulphur Dioxide	Not likely to be emitted	28 M(5)	3.0 M(8)	52 M(8)	42 G(9)	20 M(7)	53 M(6)	90 M(6)	0.11 M(7)
Hydrogen Chloride	No data	1.2 M(5)	<0.02 M(8)	32 M(8)	58 G(9)	74 M(7)	3 M(6)	14 M(6)	Not likely to be emitted
Hydrogen Fluoride	Not likely to be emitted	0.4 M(5)	<0.007 M(8)	0.34 M(8)	1 G(9)	1 M(7)	3 M(6)	2.7 M(6)	Not likely to be emitted
Volatile Organic Compounds	No data	36 M(5)	No data	11 M(8)	8 M(8)	33 M(7)	6.4 M(6)	7.6 M(6)	5.1 M(7)
1,1 – Dichloroethane	No data	No data	No data	Not likely to be em	nitted		0.66 M(6)	0.66 M(6)	Not likely to be emitted
Chloroethane	No data	No data	No data	Not likely to be em	nitted		0.26 M(6)	0.26 M(6)	Not likely to be emitted
Chlorothene	No data	No data	No data	Not likely to be em	nitted		0.28 M(6)	0.28 M(6)	Not likely to be emitted
Chlorobenzene	No data	No data	No data	Not likely to be en	nitted		0.59 M(6)	0.59 M(6)	Not likely to be emitted
Tetrachloroethene	No data	No data	0.0004 M(7)	Not likely to be em	nitted		0.98 M(6)	0.84 M(6)	Not likely to be emitted
Benzene	No data	No data	No data	Not likely to be em	nitted		0.00006 M(6)	0.00006 M(6)	0.0029 M(7)
Methane	No data	411 M(5)	No data	No data	19	No data	20,000 M(6)	19,000 M(6)	Not likely to be emitted
Cadmium	Not likely to be emitted	No data	<0.0001 M(8)	0.0069 M(8)	0.005 G(9)	0.007 M(7)	0.071 M(6)	0.071 M(6)	No data
Nickel	Not likely to be emitted	No data	<0.0003 M(8)	0.04 M(8)	0.05 M(8)	0.33 M(7)	0.0095 M(6)	0.0095 M(6)	No data
Arsenic	Not likely to be emitted	No data	<0.0005 M(8)	0.06 M(8)	0.005 M(8)	0.033 M(7)	0.0012 M(6)	0.0012 M(6)	No data
Mercury	No data	No data	<0.0006 M(8)	0.069 M(8)	0.05 M(8)	0.021 M(7)	0.0012 M(6)	0.0012 M(6)	No data
Dioxins and Furans	No data	4.0×10^{-8} M(5)	No data	$4.8 \times 10^{-8} M(8)$	$4.0 \times 10^{-7} G(9)$	2.4×10^{-6} M(7)	1.4 × 10 ⁻⁷ M(6)	$5.5 \times 10^{-8} \text{ M(6)}$	$3.8 \times 10^{-11} M(7)$
Polychlorinated Biphenyls	No data	No data	No data	No data	0.0001 M(8)	No data	No data	No data	No data
Carbon Dioxide	No data	181000 M(5)	No data	No data	1000000 G(9)	No data	300000 M(6)	200000 M(6)	1170

Note: The uncertainty associated with these estimates is set out in this Chapter. Data Pedigree:P(1-4): Poor; M(5-8): Moderate; G(9-12): Good; VG(13-16): Very Good

Introduc	tion		Review on Emi	of Information	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions	
Introduct	tion	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

 Table 2.46
 Solid residues from waste management facilities (grams per tonne of MSW)

Substance	Composting	MBT	AD	Gasification/ pyrolysis	Mass burn incineration (Bottom ash)	Mass burn incineration (APCR)	Mass burn incineration (Re-use of bottom ash)	Incineration of pre- sorted MSW
Likely route	Re-use or Iandfill	Re-use or landfill	Re-use or landfill	Re-use or landfill	Landfill	Landfill	Re-use	Landfill
Mass solid residue	500000	605000	No data	120000	180000	30000	92000	190000
Dioxins and Furans	7.2 × 10 ⁻⁹ g TEQ/T	No data	No data	No data	9.7 × 10 ⁻⁶ g TEQ/T	2.66 × 10 ⁻⁵ g TEQ/T	2.3 × 10 ⁻⁶ g TEQ/T	No data
Aluminium	No data	No data	No data	17800	5,400		1378	No data
Arsenic	1.75	No data	No data	No data	4.8	1.1	1	No data
Barium	No data	No data	No data	No data	240	No data	61	No data
Calcium	No data	No data	No data	No data	21,000	No data	5409	No data
Cadmium	0.65	No data	1.7	4	7.6	2.8	2	No data
Cobalt	No data	No data	No data	No data	3.3	0.68	<20	No data
Chromium	28	No data	95	266	56	2.2	14	No data
Copper	53	No data	118	357	390	12	100	No data
Iron	No data	No data	No data	9660	5,900	No data	1507	No data
Mercury	0.25	No data	1	No data	0.070	0.25	<0.1	No data
Potassium	No data	No data	No data	No data	1,400	No data	351	No data
Magnesium	422	No data	No data	No data	2,000	No data	523	No data
Manganese	No data	No data	No data	No data	220	11.8	57	No data
Sodium	No data	No data	No data	No data	4,300	No data	1092	No data
Nickel	6.8	No data	27	36	14	0.81	4	No data
Lead	70	No data	290	670	480	65	124	No data
Antimony	No data	No data	No data	No data	30	10	8	No data
Tin	No data	No data	No data	No data	320	19	81	No data
Titanium	No data	No data	No data	No data	690	No data	177	No data
Vanadium	No data	No data	No data	No data	14	1.6	4	No data
Zinc	117	No data	389	1700	850	No data	218	No data
Carbonate	No data	No data	No data	No data	1,800	No data	454	No data
Fluoride	No data	No data	No data	No data	36	No data	9	No data
Chlorine	No data	No data	No data	4800	790	No data	202	No data
Sulphite	No data	No data	No data	No data	42	No data	11	No data
Sulphate	No data	No data	No data	5085	4,200	No data	1065	No data
Ammonium	2.2	No data	No data	0.75	No data	No data	No data	No data
Phenol	No data	No data	No data	0.06	No data	No data	No data	No data

116

Introduction		Review on Emi	of Information ssions	Review Epidemi Researc	of ological ch	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Pub Perception Issu	antified Risks, & lic ues	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.47 Releases to sewer from waste management facilities (grams per tonne of MSW)

	Composting	MBT	AD	Mass burn incineration	Incineration of pre- sorted waste	Landfill
Suspended solids	23	No data	No data	No data	No data	No data
Mercury	0.026	No data	No data	<0.0003	<0.004	No data
Cadmium	0.0045	No data	No data	<0.003	<0.03	No data
Aniline	No data	No data	No data	No data	No data	<0.00008
Methyl tertiary butyl ether	No data	No data	No data	No data	No data	<0.00011
Chloride	680	No data	No data	No data	No data	123
Cyanide (as CN)	0.00075	No data	No data	No data	No data	<0.005
Di(2ethyl hexyl)phthalate	No data	No data	No data	No data	No data	<0.00008
Ethylbenzene	No data	No data	No data	No data	No data	<0.0008
Fluoride	No data	No data	No data	No data	No data	0.07
Methyl chlorophenoxy acetic acid	No data	No data	No data	No data	No data	<0.000008
Dichloromethane	No data	No data	No data	No data	No data	<0.00011
Nitrogen (Total)	No data	134	8	No data	No data	39
Organo-tin	No data	No data	No data	No data	No data	0.000022
Phenols	0.011	No data	No data	No data	No data	0.0023
Phosphorus	No data	No data	No data	No data	No data	0.32
Polycyclic aromatic hydrocarbons	No data	No data	No data	No data	No data	<0.00048
Nonyl phenol	No data	No data	No data	No data	No data	0.00008
Biphenyl	No data	No data	No data	No data	No data	0.000009
Месоргор	No data	No data	No data	No data	No data	0.0008
Naphthalene	No data	No data	No data	No data	No data	0.00004
Pentachlorophenol and compounds	No data	No data	No data	No data	No data	<0.000009
Toluene	No data	No data	No data	No data	No data	0.0017
Xylenes	No data	No data	No data	No data	No data	0.0031
As	No data	No data	No data	<0.006	No data	0.0007
Cr	0.023	No data	No data	<0.04	No data	0.005
Cu	0.015	No data	No data	<0.04	No data	0.001
Pb	0.033	No data	No data	<0.05	No data	<0.0054
Ni	No data	No data	No data	<0.04	No data	0.006
Zn	0.135	No data	No data	<0.02	No data	0.011
Sulphates	No data	5	No data	No data	No data	No data

117

Note: No data on remaining facilities

Introduction		Review on Emis	of Information ssions	Review Epidemi Researc	of ological h	Quantifica Conseque Emissions	tion of Health inces of	Quantificatio Environment Consequenc Emissions	n of the al es of	Context for Qu Health and Environmental Review of Publ Perception Issu	antified Risks, & lic Jes	Conclusions	
Introduction	Scope	Sources	Materials Recovery Facilities	Composting (in-vessel)	Composting (windrow)	Mechanical biological treatment	Anaerobic digestion	Gasification Pyrolysis	Inciner- ation	Small-scale incineration	Landfill	Transport -ation	Summary

Table 2.48 Releases to surface water from waste management facilities (grams per tonne of MSW)

Substance	Mass burn incineration (1 facility out of 10)	Landfill
Suspended solids	No data	No data
Mercury	<0.00012	No data
Cadmium	<0.0009	No data
Aniline	No data	<0.0000024
Methyl tertiary butyl ether	No data	<0.000024
Chloride	No data	27
Cyanide (as CN)	No data	<0.0012
Di(2ethyl hexyl)phthalate	No data	<0.0000012
Ethylbenzene	No data	<0.000047
Fluoride	No data	0.015
Methyl chlorophenoxy acetic acid	No data	<0.00000012
Dichloromethane	No data	<0.000024
Nitrogen (Total)	No data	8.6
Organo-tin	No data	0.000005
Phenols	No data	0.000007
Phosphorus	No data	0.07
Polycyclic aromatic hydrocarbons	No data	<0.00006
Nonyl phenol	No data	0.0000012
Biphenyl	No data	0.00000012
Mecoprop	No data	0.0000013
Naphthalene	No data	0.0000005
Pentachlorophenol and compounds	No data	0.000009
Toluene	No data	0.0001
Xylenes	No data	0.00021
Arsenic	<0.0055	0.000056
Chromium	<0.018	0.00082
Copper	<0.018	0.00013
Lead	<0.018	<0.0012
Nickel	<0.018	0.0011
Zinc	<0.21	0.001

Note: No data on remaining facilities

Introduction	Review of Informati on Emissions	ion Review of Epidemiologica Research	I	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Corr	posting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

3. REVIEW OF EPIDEMIOLOGICAL RESEARCH

119

Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	al	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

Summary – epidemiological research

In this chapter, the evidence for health effects arising from waste management operations is reviewed. The review considers mainly studies investigating ill-health in people living close to facilities treating or disposing of municipal solid waste (MSW). Where this information was not available, we looked more widely to consider whether useful information could be gained from studies of ill-health in workers at these kind of facilities, or from studies of facilities dealing with other types of waste.

So far as we know, there are no studies of health effects in people living near materials recycling facilities (MRFs). Some studies have been carried out on the health of workers at MRFs. These studies do not give us any reason to think that there are likely to be health effects on people living near to MRFs, but we cannot completely rule these out. Limited information on the effects of composting facilities on the health of local residents is available. This suggests that there might be a link between commercial scale composting facilities and the occurrence of respiratory and irritative symptoms in people living very close to the facilities.

We found no published information on the health effects of anaerobic digestion, pyrolysis/gasification or transport of MSW.

We looked in detail at studies of incineration facilities, and found no consistent or convincing evidence of a link between cancer and incineration. There is little evidence that emissions from incinerators make respiratory problems worse. In most cases the incinerator contributes only a small proportion to local levels of pollutants.

A recent study of landfills in the UK indicated that people living within 2 km of an active or disused landfill site in the UK may experience slightly higher rates of certain birth defects than people living further away. The study was not able to state whether the observed increases were due in part or in whole to exposure to emissions from the landfill, or to some other cause or causes. Practical difficulties in the study mean that other causes are likely to be responsible for at least some of the observed increases. A similar study found no evidence that living close to landfill sites increases the chance of getting cancer to a level that can be measured.

Finally, we estimated "dose-response functions" which were used to investigate the possible health effects of waste management operations. This is described in Chapter 4 of this report.

Introduction	Review of Informat on Emissions	ion F F	Review of Epidemiological Research	I	Quantification Consequence Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Material Fa	als Recycling acilities	Com	posting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

3.1 Glossary

Actinomycetes – a specific group of bacteria that are capable of forming very small spores.

Acute – referring to exposures and effects occurring on a relatively short timescale (e.g. hours or days).

Aeration – the process by which oxygen-rich air is supplied to compost to replace air depleted of oxygen.

Aerobic – an organism or process that requires oxygen.

Aerosol – a suspension in a gaseous medium of solid particles, liquid particles or solid and liquid particles having a negligible falling velocity.

Allergic alveolitis – condition where the lungs are allergic to fungus and other allergens which cause inflammation of the alveolar region of the deep lung.

Asthma – narrowing of the bronchial tubes, where the muscles go into spasm and the patient has difficulty breathing.

Atopy – hereditary allergic reaction which includes diseases such as hay fever, asthma etc. where there is a clear family history of these conditions.

Bacteria – a group of micro–organisms with a primitive cellular structure, in which the genetic material is not retained within an internal membrane (nucleus).

Bioaerosol - micro-organisms suspended in the air.

Biomarkers – any parameter that can be used to measure an interaction between a biological system and an environmental agent, which may be chemical, physical or biological.

Cardiovascular – of the heart and blood circulation system.

Cardiovascular disease – any disease which affects the heart or circulatory system.

Chronic – referring to exposures and effects occurring on a relatively long timescale, typically years.

Chronic pulmonary effects – Long-term disruption to the lung's ability to supply oxygenated blood to the heart.

Clinical health effect – condition causing evident symptoms.

Colorectal cancer – cancer of the colon or rectum.

Confounding factor – A condition or variable that is both a risk factor for disease and associated with an exposure of interest. This association between the exposure of interest and the confounder (a true risk factor for disease) may make it falsely appear that the exposure of interest is associated with disease.

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantification Consequen Emissions	on of Health ces of	Quantifie Environ Conseq Emissio	cation of the mental uences of ns	Cont Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

Congenital anomaly – birth defect, a malformation that exists in a person's body from birth.

Congenital malformation – an abnormal development of a body structure which is present at birth.

Cytogenetics – study of the structure and function of cells, especially chromosomes.

Dermatomycosis – skin infections caused by a fungus.

Dioxin – (abbreviation for chlorinated dibenzo-para-dioxin) – a general term that describes a group of chemicals formed by the burning of substances containing chlorine and carbon.

Eczema – non–contagious inflammation of the skin.

Endotoxin – certain (toxic) substances found within bacterial cells and which are released only following damage to cells.

Epidemiology – the scientific study of the causes, spread and distribution of disease in humans.

Epispadias – congenital defect where the urethra opens on the top of the penis and not on the end.

Exomphalos – a hernia (or rupture) present at birth which bulges at the navel.

Exposure – exposure to a chemical is the contact of that chemical with the outer boundary of the human body including the skin and openings of the body such as the mouth, nostrils, and punctures and lesions in the skin.

Exposure-response coefficient – a factor (coefficient) representing the relationship between the amount of a chemical at the absorptive surfaces of an organism and a specific adverse effect, or the incidence of an adverse effect.

Fungi – a group of micro–organisms with a more complicated cellular structure than bacteria, in which the hereditary genetic material is retained within an internal membrane, forming a nucleus.

Furan – (abbreviation for chlorinated dibenzofuran) – a member of a group of substances formed under the same conditions as dioxins.

Gasification – reaction of waste materials or residues with air and steam in the "water–gas" reaction to form hydrogen and carbon monoxide.

Gastroschisis – a fissure or split in the abdominal wall, present at birth.

Glucans – polysaccharides composed of D–glucose in either straight or branched chains with glycosidic linkages.

Haemangiosarcoma – form of malignant tumour in a blood vessel.

Hazard – the potential of an activity, object or exposure to cause harm.

Hepatobiliary cancers – a specific cancer of the liver.



Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revie Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

Hypospadias – congenital defect of the wall of the male urethra or the vagina, so that the opening occurs on the under side of the penis or in the vagina.

Immunoglobulin – antibody, a protein produced in blood plasma as protection against infection.

Impact – a measure of the effect of an activity, object or exposure upon a receptor.

Leukaemia – any of several malignant diseases in which an abnormal number of leucocytes (white blood cells) form in the blood.

Municipal solid waste (MSW) - solid waste collected by, or on behalf of local authorities.

Mycotoxins -toxic substances produced by fungi.

Nasal lavage – a procedure in which washings are taken from the inside of the nose.

Neural tube – tube lined with ectodermal cells running the length of an embryo, which develops into the brain and spinal cord.

Neural tube defects – congenital defect which occurs when the edges of the neural tube do not close up properly (e.g. spina bifida).

Non-Hodgkin's lymphomas – tumours arising from lymphoid tissue that are not a result of Hodgkin's disease.

 PM_{10} – mass concentration of particulate matter collected by a sampler with a 50% cut-point at an aerodynamic particle diameter of 10 μ m; mostly particles with aerodynamic diameter of 10 μ m or less.

 $PM_{2.5}$ – mass concentration of particulate matter collected by a sampler with a 50% cut-point at an aerodynamic particle diameter of 2.5 μ m, mostly particles with aerodynamic diameter of 2.5 μ m or less.

Pneumonitis – inflammation of the lungs.

Polychlorinated biphenyls (PCB) – a chloro–biphenyl organic pollutant produced in various industries.

Polycyclic aromatic hydrocarbons (PAHs) – hydrocarbon compounds with multiple fused benzene rings. PAHs are typical components of asphalts, fuels, oils, and greases.

Pyoderma – eruption of pus in the skin.

Pyrolysis – heating of materials such as municipal solid waste in the absence of oxygen.

Renal - of the kidneys.

Rhinitis – inflammation of the mucous membrane of the nose.

Risk – the likelihood that a hazard will actually cause harm.

Introduction	Review of Informat on Emissions	ion	Review of Epidemiological Research	I	Quantificatio Consequent Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revie Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Mater	rials Recycling Facilities	Com	posting	Other W Disposal C	'aste Options	Incineration	ı	Landfill	Tables

Risk assessment – an evaluation, often quantitative in nature, of the level of risk associated with an activity, object or exposure.

Sarcoma – cancer arising in bone, connective tissue or muscle.

Sinusitis – inflammation of the mucous membrane in the sinuses.

Spirometry – measurement of the vital capacity of the lungs.

Statistical power –the power of a study to detect a true effect – i.e., the probability that a statistical significance test will reject the null hypothesis when the null hypothesis is not true.

Thermophilic – the process or organism that occurs at temperatures above 45° C.

Toxicity – degree to which a substance is poisonous or harmful.

Tracheo-bronchitis - inflammation of both the trachea and the bronchi.

Volatile organic compounds (VOCs) – a group of organic compounds that volatilise easily at ambient temperatures. Some VOCs are linked with environmental effects such as photochemical smog and ozone depletion, and some are toxic and/or carcinogenic.

Windrow – elongated pile of composting material.

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

3.2 Introduction

Waste management is a very large scale activity which inevitably has consequences for human health and the environment. At the very least it involves transporting waste materials, although the environmental consequences of transport are not considered within this study. The various waste management processes such as landfill and incineration are very different in character and give rise to different kinds of human health hazards.

Domestic waste is the main component of municipal solid waste and this can contain hazardous substances such as pesticides used within the home. If such substances are volatile then it is likely that they will be released from a landfill. Incineration may destroy such substances but combustion itself is well known to create toxic substances such as sulphur dioxide, oxides of nitrogen, dioxins and furans. Composting can also generate hazardous substances – for example, some of the micro-organisms which flourish in the composting process are able to release spores with allergenic properties which can stimulate or exacerbate respiratory diseases. Even recycling processes are not without risk. These may well involve the expenditure of energy and consequent release of combustion gases and/or produce contaminated wash waters.

This chapter is concerned with reviewing the available evidence on the effects on local populations of waste management activities. In the case of composting and MRFs, data on community residents is sparse or even non-existent, and in these cases information on effects on workforces is also reviewed. In relation to incinerators and landfills, two approaches are adopted. Firstly, available epidemiological studies are reviewed and secondly, putative health effects arising from atmospheric emissions are calculated from knowledge of the emissions and their effects on health. In the case of epidemiological studies of incinerators, the results of studies involving different kinds of incinerators have been included, since the nature of toxic pollutants emitted from a clinical waste incinerator or even a hazardous waste incinerator is likely to be broadly similar to that of a municipal waste incinerator. In the case of landfill, however, because of the capacity of landfill to cause public exposure to substances disposed in the landfill rather than, for example, products of combustion of those substances, detailed review is carried out only on studies which focus on landfill accepting only municipal solid waste, and therefore research on hazardous waste landfills and co-disposal sites is not considered directly relevant to evaluating the health effects of landfill.

When considering the health effects of an individual facility, it is important to take account of the local circumstances and any evidence of sensitivity of local residents to the health effects of concern.

References which are published in the peer reviewed literature are marked in **bold**. References which have been reviewed by other means (e.g. Environment Agency research reports, or information published by reputable governmental bodies) are marked in *italics*. Other references are marked in normal type.

3.2.1 Epidemiological studies

The term epidemiology describes the study of diseases in populations, as opposed to the clinical study of individual patients. Thus, epidemiology tends to be concerned with the patterns of disease in society, from which knowledge can be gained of the causes and spread of disease and measures which can be used to control disease. Descriptive epidemiology involves determining the prevalence



Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

rates of different diseases, patterns with respect to factors such as age, sex, ethnicity and socio-economic status, as well as the influence of weather and season upon disease occurrence. It can lead to specific kinds of epidemiological study such as cohort, case control and intervention studies which are used to elucidate factors influencing diseases within society.

Research in environmental epidemiology typically seeks to link exposure to a toxic substance with adverse health effects in the general population. To do so implies that there must be some measure of exposure and also some measure of health outcome. Neither is necessarily as straightforward as it sounds.

Measuring exposure

In the case of municipal waste, disposal activities can generate a range of potential emissions of substances causing contamination of air, soil and water media. Generally speaking, epidemiological studies have not sought to measure human exposure to specific pollutants. Indeed, rather few have made any measure of pollutant concentrations at all. Given the complexity of the pollutant mixture, the possibility of exposure through multiple pathways and the generally non-specific health outcomes which could be attributed to more than one pollutant, it is arguable that there would be rather little to be gained from trying to measure exposure directly. It would be immensely complex and expensive to evaluate experimentally the exposure of large numbers of people to a wide range of pollutants through multiple environmental media. Consequently, greatly simplified approaches have been used, and the majority of studies have used simple proximity of residence to the source as a surrogate for the level of exposure.

In the most comprehensive study of populations living close to landfill sites, people residing within two kilometres of a landfill were defined as exposed whilst those residing at a greater distance from any landfill were considered as unexposed. Clearly, this is a gross over-simplification but probably justifiable in view of the difficulties of making a more sophisticated assessment. In the slightly more sophisticated studies, concentric circles may be constructed around a point source of emissions such as an incinerator, and a gradient in exposure assumed between the inner-most and outer-most circles. One criticism which applies to the majority of studies in which the airborne exposure route is likely to be most important is that the non-uniform distribution of wind directions is not taken account of despite the fact that it may have a strong influence upon population exposure. The effect of these uncertainties is to lead to misclassification of exposure in which some of the people classified as exposed actually have low exposures and *vice versa*. Exposure misclassification in this way reduces the overall confidence that an observed effect is real.

Measuring health outcomes

Evaluating health outcomes is usually most straightforward in regard of cancers since the cancer registration process in the UK (and in many other countries) records the point of residence of those recorded as contracting the disease. It is therefore possible to analyse retrospectively the incidence of cancer in relation to some surrogate for pollutant exposure. In the case of most other illnesses such as respiratory disease, there is no comparable registration process and routine health service data such as hospital admissions cannot be disaggregated according to location of residence. Epidemiological studies of these outcomes must depend upon self-reported symptoms, which are recognised to be frequently unreliable, or objective measures such as lung function which cannot be measured on large numbers of people. This consequently reduces the power of the study to identify an effect.

Introduction	Review of Informati on Emissions	ion Review of Epidemiol Research	gical	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycl Facilities	ng Cor	nposting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

Epidemiological methods

Environmental epidemiology seeks to investigate connections between environmental factors such as pollutant exposures and adverse effects on health. There are a number of available methods but two have been applied most commonly in the areas addressed by this study.

Spatial analysis methods depend primarily upon comparing rates of disease amongst exposed and unexposed populations. Most commonly distance from source is the measure of exposure, although other ways of describing exposure are sometimes used. If the population living within a short distance of a pollutant source (after controlling for confounding factors; see below) shows a higher frequency of disease outcomes per head of population than occurs in those living at a slightly greater distance, then pollutant exposure due to the source in question may be the cause and further investigation is warranted. A statistical association alone between exposure and disease does not establish causality and there are criteria which are commonly applied to form a view on whether an association implies a causal connection. Clearly, if the effect detected is one that is known to be associated with a pollutant emitted from the source, this adds considerably to the strength of evidence for causality but a number of other criteria also need to be met.

There is no standard list of criteria for establishing whether a statistical association between a factor such as pollutant exposure and the occurrence of disease is causal, i.e. whether the pollutant is the cause of the disease. However, factors typically applied in making a judgement, and which are implicit in most of the studies reviewed and in our assessment of them are as follows:

- Strength of association: A large magnitude of effect and high statistical significance is likely to be far more convincing than a small effect of marginal statistical significance.
- Time sequence: The occurrence of the disease must follow the exposure in time. If the disease precedes the exposure, then causality is highly improbable.
- Distribution of the disease: If the disease varies in space and time in the same manner as the causal factor (allowing for possible latency periods), then causality is much more likely.
- Exposure-response gradient: It would be expected that large exposures are associated with more cases of consequent disease. Areas highly exposed to pollutants would be expected to show a greater prevalence of consequent disease than those with a low exposure.
- Consistency and coherence: If a number of studies, for example in different places, showed the same relationship between exposure and disease, this would be referred to as consistency and would add weight to arguments for causality. If evidence from different kinds of studies also shows the same kind of association between exposure and disease, this coherence would also be taken as evidence favouring causality.
- Biological plausibility: The association between the disease and exposure to the suspected causal agent should be consistent with the known biological activity of the suspected agent.
- Experimental models: There are a range of experimental models such as laboratory animals which can be used to evaluate the consequences of



Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	erials Recycling Facilities Cor		Other W Disposal C	/aste Options	Incineration	I	Landfill	Tables

exposure to chemicals. If the results of such experiments are consistent with the statistical associations established through epidemiology, the case for causality is strengthened.

The second main route of investigation is the case control study. Such studies deal with individuals who have already developed disease which may be associated with emissions from the source in question. These so-called cases are then matched with controls of a similar age and socio-economic status who are not suffering from the disease. Risk factors such as location of residence are then established, and if a larger proportion of the cases as opposed to the geographically randomly selected controls live within the exposed zone, a tentative conclusion may be drawn that the source investigated may be responsible for the observed adverse effects.

Two points should be emphasised for those unfamiliar with the interpretation of epidemiological research. The first is that many of these studies looked at a wide range of adverse health outcomes and that even at a 95% confidence level, a small proportion of results will be positive by chance (in this case 5% on average). Thus, if for example 20 adverse health outcomes were investigated, on average one will be positive purely by chance. In examining the outcomes of such studies, it is necessary to look for evidence of consistency between studies rather than taking a single statistically significant positive finding as necessarily indicative of an effect. The second very important point is that a statistically significant finding in a study is only a demonstration of a statistical association. There are a range of other criteria which should be used to infer or reject causality of the association, such as the magnitude of the observed association.

Three other issues add to the considerable complexity of interpreting the results of epidemiological studies.

Confounding

Epidemiological studies seek initially to establish statistical relationships between a disease vector and the disease itself. This does not necessarily imply causation since the statistical correlation may arise by chance or because one or more further variables are correlated with both the exposure and the health outcome. One such variable is socio-economic status. People in higher socio-economic groups with greater purchasing power rarely choose to live close to waste disposal facilities and it is therefore the poorer members of the community, who are also statistically more liable to contract most diseases for reasons unconnected with pollutant exposure, who live in the zones closest to emissions from industry and waste management operations. Thus, even in the absence of a pollutant effect, one might well expect to find a correlation between proximity to an incinerator, for example, and a range of diseases, including for example lung cancer. The researchers in good quality studies therefore control for (i.e. adjust the results of the study to allow for) confounding by socio-economic status, but it is unlikely that any control is perfect and hence there may be some residual confounding which leads to an apparent effect which may well not be real. Long-ceased industrial operations may also leave local contamination which may be a cause of disease, and industrial activities are often sited in close proximity, making allocation of responsibility for effects to a given source problematic.

Whilst socio-economic status is often the most important confounding factor, there are many other sources of confounding such as age, gender, ethnicity, access to healthcare, smoking prevalence, occupation etc. Ideally, an epidemiological study will control for all of these variables, but in practice this is rarely practicable and it is unusual to control for other than socio-economic status. The consequence is that

Introduction	Review of Informat on Emissions	on Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

there may be residual confounding which could lead to a false result, either positive or negative.

Cancer latency

There is considerable evidence from studies of adults that cancer due to pollutant exposure may not manifest itself for many years after that exposure. A classic example is cigarette smoking and lung cancer, where a change in lung cancer incidence typically lags a change in smoking frequency in the population by 20 years or more. In the case of other cancers which occur in children such as leukaemia, clearly the latency period cannot be so long. Most epidemiological studies of cancer only have access to information on the location of residence at the time that the cancer was recognised. This may of course be different from the location at an earlier time when a key pollutant exposure may have occurred. In a situation where latency periods probably differ between different cancers, and in any case are not known with certainty, it is not possible to allow readily for latency periods in the analysis of geographically resolved data. Thus, the occurrence of latency and population migration will weaken any relationships between proximity to a pollution source and the occurrence of cancers, making it more difficult to identify possible causal relationships between pollutant exposures and cancer. In the case of teratogenic effects of chemicals (i.e. those relating to birth deformities), latency periods are not well known. These are likely to be relatively short if it is exposure of the foetus rather than the parent which is important, although there is evidence that this may not always be the case.

Statistical power

The above term describes the power of an epidemiological study to identify an association at a given level of statistical significance. It depends on a number of factors, most importantly the size of the population studied. For this reason, the most powerful studies have aggregated large populations from multiple sites (e.g. landfills), rather than investigating a single site. It should be stressed that a study indicating no significant excess risk may do so because of inadequate statistical power. This should not, therefore, be interpreted as necessarily implying a complete absence of risk; rather it might be consistent with a low level of risk.

Introduction	Review of Informat on Emissions	Review of Information in Emissions Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities		nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

3.3 Materials Recycling Facilities

3.3.1 Introduction

A materials recycling facility (also known as a materials recovery facility or materials recycling factory) is defined as a central operation where source segregated, dry recyclable materials are sorted mechanically or manually to market specifications for processing into secondary materials (*Gladding, 2002*). As implied by this definition, waste material entering an MRF has normally been subject to some pre-segregation by the householder, but further sorting is required which may involve machinery or may be by human hand. Whichever, it is likely that human operators come into close contact with the waste.

The hazard which probably presents the greatest health risk relates to biological materials, and particularly bioaerosol. The associated risks are very similar to those occurring in a composting plant as outlined in Section 4, although likely to be of lower magnitude if mainly dry recyclables are handled.

Unlike the composting plant, there are also significant chemical and physical hazards to the worker in the MRF, and those chemical hazards including exposure to vapours and suspended particulate matter may extend outside of the plant. *Gladding (2002)* reports measurements of airborne contaminants within plants where unseparated waste is sorted, and in addition to high concentrations of airborne micro-organisms, there are reports of elevated concentrations of toxic trace metals. Such plants do not currently operate in the UK, although a facility is currently under development.

3.3.2 Review of health effects studies

There are no epidemiological studies of populations living near MRFs. To enable the potential health effects in local populations to be assessed, we have considered studies of worker exposures. Relevant studies of the atmosphere and/or health of workers in MRF plants appear in Tables 3.1 and 3.2. According to *Gladding (2002)*, the most heavily investigated mixed wastes facility was a sorting plant receiving up to 10,000 tonnes per annum of mixed household and industrial waste. Of 15 exposed operatives, five exhibited symptoms of asthma whilst others presented with flu-like symptoms (possibly allergic alveolitis), eye and skin irritation, fatigue and occasional nausea (Malmros, 1988). A further study of operatives from the same plant (Sigsgaard, 1990) showed that eight operatives became ill within seven months of starting work. In total, nine cases of occupational disease among the original 15 exposed operatives were reported.

A subsequent study in Denmark examined the health of 750 operatives in textile mills, recycling plants, a wet paper producing plant and a water supply plant (**Sigsgaard, 1993**). The tests applied included lung function testing, peak flow monitoring, skin prick test, serological tests and interviews. Viable airborne bacteria and fungi were collected. Within these industries a negative association was found between different markers of atopy and increasing levels of endotoxin, which implies a healthy worker selection due to operatives leaving recycling after a short period of employment because of asthma symptoms. When odds ratios were examined, they showed that chest tightness and organic dust toxic syndrome was significantly elevated amongst waste handling operatives who also had significantly increased rates of work-related chest tightness, influenza, feeling of fever, and mucous membrane irritative symptoms. Gastro-intestinal symptoms such as nausea, work-related vomiting and diarrhoea were also more often reported

Introduction	Review of Informat on Emissions	tion	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantifi Environ Conseq Emissio	Quantification of the Environmental Consequences of Emissions		ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Mater	rials Recycling Facilities Com		posting	posting Other W Disposal C		Incineration	1	Landfill	Tables

amongst waste handling operatives than amongst controls. A study conducted in the United States conducted air sampling in six MRFs (*IEERR, 1995*). Measurements of silica and trace metals including arsenic, aluminium, chromium, lead, nickel and mercury were low compared to occupational standards. Similarly, measurements of metals in downwind communities as well as PCBs and pesticides show little evidence of elevation due to the MRF. Some elevation in concentrations of total suspended particulates and PM₁₀ was evident in the measurements. In the case of bioaerosol, no significant difference between upwind and downwind concentrations measurements was evident. The study concluded that there was little evidence that MRFs posed a significant threat to public health or the environment, but drew attention to the possible problems associated with bioaerosols and the lack of widely adopted occupational exposure limits for bioaerosols.

Marth *et al.* (1999) studied the health of 256 workers from manual sorting facilities over a period of three years. This showed a decrease in lung function and an increase in total Immunoglobulin E in exposed workers indicative of allergic sensitisation to atmospheric exposures.

Gladding (2002) reports on the European BIOMED No. 2 project which was designed to provide information on physical and chemical hazards in MRFs and health effects of measured bioaerosol exposures. Work was conducted in 11 MRFs handling a mixture of household and commercial waste materials. The various MRFs were divided into three exposure groups according to concentrations of total dust, endotoxin and glucan. The study took account of the prevalence of smokers amongst the workers as well as age, sex, presence of chronic disease and length of time working at an MRF. The results for the association between symptoms and exposure category appear in Table 3.3. In the case of some of the symptoms, gradients are clearly observable between the symptom and the level of exposure. These results have been converted into odds ratios in Table 3.4 and the significant associations are indicated with an asterisk. This is the first clear demonstration of an exposure-response gradient for workers in an MRF. However, there are weaknesses to the study, particularly that the health survey was based on a selfreporting of certain symptoms by questionnaire. This method is notoriously subject to bias where the purpose of the research is known to the respondents. In this case, this seems less likely to lead to bias since the workers were not aware of the exposure levels. Indeed, the survey may under-estimate the effects of exposure to bioaerosol since there is evidence that individuals with atopic diseases are significantly under-represented in compost workers and that healthy worker selection tends to eliminate the more sensitive individuals from exposure.

3.3.3 Development of quantitative health impact functions

Epidemiological studies of the health of workers in materials recycling facilities are very few in number, and only one has demonstrated clear exposure-response gradients for a range of conditions amongst the workers. The concentrations of suspended particles, endotoxin and glucan to which those workers were exposed were far in excess of those to which the general public are exposed in the vicinity of an MRF. There do not appear to be any reported studies of the health of local populations in the vicinity of MRFs and therefore it is not possible to recommend any quantitative health impact functions. It should not, however, be assumed that effects on the general population are negligible, as the shape of the exposure-response functions is unknown and the general public is likely to include individuals of far greater susceptibility than the workforce.

For Materials Recovery Facilities, as noted above, the most important emissions from such sites in public health terms are likely to be bioaerosols. As such, the



Introduction	Review of Informat on Emissions	mation Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

quantitative exposure-response functions derived for composting facilities as outlined in Tables 3.7 and 3.8 below may be applicable.

Introduction	Review of Informat on Emissions	ion Revie Epide Resea	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Re Facilitie	erials Recycling Facilities Com		posting	Other W Disposal C	'aste Options	Incineration	ı	Landfill	Tables

3.4 Composting

3.4.1 Introduction

The composting process may be defined as the controlled biological decomposition and stabilisation of organic substrates, such as vegetable, plant and some food wastes, under conditions that are predominantly aerobic and that allow the development of thermophilic temperatures as a result of biologically produced heat (**Swan et al., 2002**). It results in a final product typically referred to as "compost" that has been sanitised and stabilised, is high in humic substances and can be beneficially applied to land. Composting is now employed as a treatment process for a wide range of organic substrates such as municipal solid wastes, sewage sludges and agricultural and industrial bio-products. Actively composting materials or finished composts have been shown to degrade a wide range of organic pollutants and are thus used in the bioremediation of contaminated soils.

Organic materials received at a composting facility require pre-processing, involving four main activities, namely shredding, mixing different feedstocks together to improve homogeneity and adjust the carbon to nitrogen ratio and/or moisture content, adding of water to optimise moisture content and removing contaminants. The pre-processed waste then goes into one of a number of large scale composting processes which include windrow systems, aerated static piles, in-vessel systems and vermicomposting (**Swan et al., 2002**). Following the composting process, post-processing takes place in order to screen out large particles and blend with other materials.

The composting process is specifically managed so as to encourage the growth of thermophilic heterotrophic aerobic organisms which can survive the high temperatures generated in the composting process and produce the enzymes needed to promote biodegradation. Whenever composting materials are moved around such as during shredding, turning and screening, the formation of a bioaerosol from the micro-organisms is an inevitable consequence. Whilst during optimal management, the composting process generates temperatures sufficient to destroy most pathogenic bacteria, these may still survive in any regions of the compost which do not reach an adequate temperature and can also be subject to aerosolisation (i.e. becoming suspended in the air) during disturbance of the compost. It is also in the nature of biodegradation processes that large complex molecules are broken down into much smaller organic molecules which are far more volatile. Thus composting releases volatile organic compounds which may also present a risk to health. The extent to which these result in exposure of the general population depends on the degree of enclosure of the operations.

Specific components of the bioaerosol generated during composting are as follows:

- Fungi these proliferate during the composting process and are of concern since some are allergenic. Amongst these is Aspergillus fumigatus which is an opportunistic pathogen which can cause aspergillosis in immunocompromised subjects.
- Bacteria these include a wide range of both Gram-negative and Gram-positive organisms, many of which arrive in the feedstock. These include faecal coliform organisms which should be destroyed by the elevated temperatures unless inadequate turning leads to temperature stratification. Another bacterium of concern, which can be present in compost is *Leptospira* which is the causative agent of Weil's disease.



Introduction	Review of Informat on Emissions	nation Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Facilities Com		Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

- Actinomycetes these are filamentous Gram-positive bacteria, some of which are thermophilic and thrive in wet compost. They are of concern since the thermophilic actinomycetes species are recognised respiratory allergens which produce large numbers of very small spores (1-3 µm diameter) capable of deep penetration in the human lung.
- Endotoxin this is a term given to fragments of the bacterial cell wall from all Gram- negative bacteria and some blue-green algae. Endotoxin is present in compost as a consequence of the presence of Gram-negative bacteria as well as being a component of some organic dusts which may contribute to feedstocks. Endotoxin may cause both short-term and long-term illness by inhalation.
- Mycotoxins these are non-volatile low molecular weight secondary metabolites produced by fungi. Most commonly encountered through ingestion, mycotoxins can be carcinogenic, neurotoxic and teratogenic, and may also contribute to occupational lung disease in workers exposed to organic dusts (Swan et al., 2002).
- Glucans these are polymeric species of glucose found in the cell walls of fungi, some bacteria and plants. (1 → 3)-β-D-glucan is a potent inflammatory agent which has been associated with an increased prevalence of atopy, decrease in lung function and adverse respiratory health effects in the indoor and occupational environment. There is evidence that (1 → 3)-β- D-glucans may enhance pre-existing inflammation in human subjects.

3.4.2 Review of health effects studies of composting

According to **Swan** *et al.* (2002) the effects of exposure to organic dust (containing the bioaerosol components listed above) on respiratory health may lead to a number of distinct identifiable conditions as follows:

- Allergic rhinitis and asthma: these inflammatory conditions are well known consequences of exposure to allergens present in organic dusts.
- Chronic bronchitis and chronic obstructive pulmonary disease (COPD): these are inflammatory diseases of the respiratory system in which long-term rather than intermittent changes in the lung cause obstruction of air exchange. There is evidence that airborne bacterial endotoxins and other factors may be one causative factor in these diseases.
- Extrinsic allergic alveolitis and granulomatous pneumonitis: these are specific inflammatory reactions of the deep lung leading to acute symptoms of chills, fever, dry cough, malaise and increasing breathlessness with the long-term possibility of permanent lung damage.
- Toxic pneumonitis or organic dust toxic syndrome: this is an acute illness occurring during or shortly after high exposures to airborne dust leading to influenza-type symptoms.

Whilst many measurements of airborne concentrations of organisms have been made within and in the vicinity of composting plants (e.g. *Wheeler et al., 2001*), which give ample evidence for a hazard especially to composting workers, there have been very few studies of health effects from which any quantitative indication of risk can be derived.

Introduction	Review of Informat on Emissions	ion	Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Mater	rials Recycling Facilities Com		posting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

There are few epidemiological studies of populations living near composting facilities. To enable the potential health effects in local populations to be assessed, we have also considered studies of the health of workers in composting facilities, for which more studies are available (see Table 3.5). Care must be taken in interpreting the results of studies of workers' health. There is a well known effect known as the "healthy worker effect" in which a workforce becomes a self-selecting population. Those who suffer ill-health as a result of their work tend to leave and seek other forms of employment whilst those who are more resistant to the effects of occupational exposures are more likely to continue in that line of employment.

Bunger et al. (2000) examined the health of 58 compost workers, 53 bio-waste collectors and 40 controls. The compost workers were found to have significantly more disease of the airways and skin than the control subjects. Disease conditions showing an excess amongst the compost workers were tracho-bronchitis, mucous membrane irritation, sinusitis, eczema, dermatomycosis, pyoderma, nausea and ear Twenty compost workers had one or more increased antibody inflammation. concentrations of relevance to their exposures compared with only three bio-waste collectors and one control. This study provides strong evidence that compost workers develop more effective immune systems as a result of their occupational exposures. Douwes et al. (1997, 2000) studied 14 Dutch compost workers and ten They measured personal exposure to dust, endotoxin, glucans, total controls. fungi, total bacteria and Gram-negative bacteria. The workers were found to suffer an excess of acute and sub-chronic non-immune or type III allergic inflammation in the upper airways which the authors suggested was induced by exposure to agents such as endotoxins and $\beta(1\rightarrow 3)$ -glucans.

Not all studies have shown positive results. **Marth** *et al.* (1997) examined the health of 137 employees at different waste handling facilities including two composting facilities and three waste sorting plants by use of a medical examination questionnaire and immunoglobulin E measurements. They report no statistically significant increase of allergic diseases amongst those occupationally exposed to waste.

There is evidence of gastric infections in composting employees. **Ivens et al.** (1997a) report an association between fungal exposure and diarrhoea in composting employees. In subsequent papers, **Ivens et al.** (1997b, 1999) found exposure-response relationships between nausea and endotoxin exposure, and between diarrhoea and exposure to both endotoxins and viable fungi in a group of waste collectors with very high exposure levels.

Studies of health in the general public around composting sites are even fewer. **Cobb et al. (1995)** compared symptoms of ill-health in those living within 900 metres of a site processing mushroom compost with a group with no exposure to compost, finding no demonstrable excess of disease in the exposed group. **Browne et al. (2001)** report on a study of health in a population living close to a grass and leaf composting plant in the US. This depended on self-reported symptoms, with 63 subjects living near the site and 82 controls keeping a symptom diary. Bioaerosol concentrations were measured at fixed locations around the site and elevated concentrations were shown to be associated with winds from the composting plant. No significant increase in allergy and asthma symptom prevalence was detectable in people living near the site, although there was an association of increases in respiratory and irritative symptoms with ragweed pollen, ozone and temperature. These are unlikely to be attributable to the composting activity.

The latter view is supported by the most recent research, summarised below.

Introduction	Review of Informat on Emissions	ion	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materi F	Facilities Com		oposting Other W Disposal C		Vaste Incineration		ı	Landfill	Tables

A recently published study by **Herr et al. (2003)** provides the most convincing evidence to date of an effect of bioaerosol emitted by a composting plant on respiratory health of community residents. The study used distance between home and the composting site as well as numbers of colony-forming units of bioaerosol as exposure measures. An analysis of prevalence of self-reported health complaints amongst residents in the neighbourhood showed a significantly elevated risk for a number of health complaints including bronchitis, frequency of colds and measures of eye irritation and general health. Whilst the use of self-reported symptoms frequently leads to bias in epidemiological studies, the authors tested this possibility through including odour annoyance as a question in their doctoradministered questionnaire. The results showed that odour annoyance, which might be expected to be a strong bias on self-reported complaints had no influence on the reporting of irritative airway complaints and this therefore did not appear to be a confounder.

Volatile organic compounds from composting

As mentioned above, release of volatile organic compounds from composting facilities may present some risk to health. **Eitzer (1995)** reports concentrations of some 36 volatile organic compounds measured within the atmospheres of eight composting facilities ranging in size from 5-10 up to 660 tonnes per day design capacity. Maximum concentrations for some species can be very considerable. For example, the highest reported concentrations of trichlorofluoromethane was 900 mg m⁻³. However, it is only maximum observed concentrations and not long-term averages which were reported and there is no indication of the extent of release to the external atmosphere and therefore atmospheric concentrations in the downwind environment. **Wheeler et al. (2001)** also measured VOC concentrations at four composting plants. Samples for VOC analysis were taken at points upwind and downwind of the operations. Results are reported for 10 VOC, which in the main represent no more than a few µgm⁻³, and are not appreciably elevated above typical urban concentrations. No evaluation of health effects related to exposure to the VOCs was reported.

3.4.3 Development of quantitative health impact functions

As outlined above, none of the epidemiological work carried out on occupationally exposed workers provides exposure-response functions useable for quantitative health effects evaluation. *Wheeler et al. (2001)* in a report entitled "Health Effects of Composting" addressed this issue through reviewing reference values (i.e. analogous to air quality standards) for bioaerosols. Whilst they were able to identify suggested reference levels for certain components of bioaerosol, these were in the form of occupational exposure limits which were not intended for application to the general public. Unfortunately, no exposure-response functions of a quantitative nature appear to have been developed in any of the published research on occupational exposure. However, the recent work of **Herr et al. (2003)** provides relative risk data which can be used to develop effects estimates.

Wheeler et al. (2001) also compared measured concentrations of VOCs with air quality standards from the UK and reference values from the USA. They were unable to identify with confidence those VOCs which were emitted from the composting process as opposed for example to emissions from passing road traffic. Concentrations of VOCs reported in an Austrian study of mechanical biological treatment plants that treat whole waste (Lahl et al., 1998) revealed very low concentrations of the majority of VOCs (see Section 6.11). The concentrations reported appear in Table 3.6. Of these compounds, only one, benzene, is a genotoxic carcinogen for which a lifetime unit risk factor of 6 x 10^{-6} (µg m⁻³)⁻¹ is recommended by the World Health Organisation (*WHO*, 2000).


Introduction	Review of Informat on Emissions	ion Ri Ej	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Fac	Is Recycling icilities	Com	posting	Other W Disposal C	'aste Options	Incineration	1	Landfill	Tables

For composting, as noted above, the literature provides little basis for quantitative exposure-response functions for the bioaerosol releases which present the most important hazard from composting sites. However, the recent study of Herr et al. (2003) gives odds ratios which relate both to distance from the source and bioaerosol concentrations in air. Tables 3.7 and 3.8 list appropriate functions derived from this study. While these were not used further in this study, they may prove useful for site specific application, or for further investigation of the health effects of composting.

Table 3.7 Adjusted odds ratios for disease due to bioaerosol exposure adjacent to a composting plant (derived from Herr et al., 2003)

Reported health complaint	Odds ratio ¹
Bronchitis	3.59
Waking up due to coughing	6.59
Coughing on rising during the day	3.18
Excessive tiredness	4.27
Current medication intake	2.64

Note 1: Adjusted odds ratios for residence 150-220 metres from composting site relative to a background location; adjusted for residential odour annoyance, duration of present residency > 5 years, composting in own garden, separate collection of organic household waste, distance of home to busy road < 50 m, age, gender and passive smoke exposure.

Reported health complaint	Odds ratio* for bioaerosol > 10 ⁵ CFU m ⁻³	Odds ratio* for duration of residency > 5 years
Frequency of colds > 5 x / year	n.s. **	4.72
Bronchitis	3.02	2.91
Waking up due to coughing	2.70	2.51
Wheezing	n.s.	2.95
Shortness of breath at rest	3.99	n.s.
Coughing on rising or during the day [†]	2.67	n.s.
Shortness of breath after exertion	4.23	n.s.
Itching eyes > 10 x / year	n.s.	2.85
Smarting eyes > 10 x / year	2.44	2.42
Nausea or vomiting > 5 x / year	n.s.	4.10
Excessive tiredness	2.80	n.s.
Shivering	4.63	3.67

Table 3.8 Adjusted odds ratios for health effects of exposure to bioaerosol for concentration of bioaerosol and duration of residency

* Odds ratio compared to those living > 400 metres from the site adjusted for odour annoyance, period of residence > 5 years and age. Odds ratios provided for population experiencing background concentrations of outdoor bioaerosol > 105 CFU m-3 air, and duration of present residency with a neighbouring composting site.

** n.s. = not significant

† Criteria of the World Health Organisation for chronic bronchitis

Introduction	Review of Informat on Emissions	Review of Information on Emissions Review of Epidemiological Research Materials Recycling		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	aste/ Aptions	Incineration	n	Landfill	Tables

3.5 Other Waste Disposal Options

The abstract databases were also searched for other waste disposal operations and specifically for those studies looking at the impact of these operations on human health. The search included:

- Anaerobic digestion;
- Pyrolysis and gasification;
- Transportation.

However, the search yielded no additional references describing epidemiological studies of the impacts that these disposal operations have on human health.

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	Review of Epidemiological Research terials Recycling		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n 🗌	Landfill	Tables

3.6 Incineration

3.6.1 Introduction

Incineration involves combustion of waste at high temperatures for a sustained period achieving a substantial reduction in the volume of waste and effectively destroying pathogenic biological organisms. Concerns over incineration relate to the by-products of the combustion process, most particularly the emissions to atmosphere, which are quantified in Chapter 2 of this report. Outputs from incinerators include:

- energy, as heat and power;
- furnace bottom ash which contains a large proportion of the non-volatile and non-combustible material such as metals contained in the original waste stream;
- air pollution control residues, which contain a significant enrichment of toxic pollutants and therefore are generally treated as a hazardous waste;
- emissions of gaseous combustion productions; and
- in some cases cooling water discharges.

3.6.2 Review of health effects studies

Modern, well-managed incinerators can be an effective means of reducing and disposing of waste materials. However the by-products of the combustion process may contain hazardous or toxic pollutants and emissions will add to background pollution levels. As a result, there is often considerable public concern over the possible health effects of living near to incinerators processing hazardous, clinical or municipal waste. This report presents a systematic review of epidemiological studies of the public health effects of waste incinerators.

There is no doubt that air pollution (from all sources) can have an adverse effect on the health of susceptible people (*i.e.* young children, the elderly and particularly those with pre-existing respiratory disease). Recent work in the UK by the Committee on the Medical Effects of Air Pollutants (COMEAP) has demonstrated that exposure to air pollution can bring forward death in patients with severe preexisting disease, although the degree of life shortening is typically of the order of a few weeks at most per individual. However, there is currently little convincing evidence that ambient levels of air pollution cause adverse health effects in healthy people.

Whilst incinerators generate a considerable amount of public concern, there have been few published epidemiological studies that examine the health of communities living in close proximity to them. The majority of published studies concentrate on the effects of exposure to emissions from the older generation of incinerators which were phased out in the UK after the introduction of stricter emission controls implemented through the Integrated Pollution Control regime. This is inevitable, in view of the latency period associated with most cancers. The level of public exposure from such facilities was substantially higher than occurs from modern incinerators.

Introduction	Review of Informat on Emissions	mation Review of Epidemiological Research Materials Recycling		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

Most of the epidemiological studies of possible health outcomes in populations living close to incinerators have not given clear indications of the presence or absence of an effect. Of necessity, many of the studies examining possible health effects are retrospective and employ routinely collected data such as cancer registrations, birth and death records. As noted in Section 1, whilst such observational studies can provide evidence of association between a health outcome and an environmental pollutant, they cannot, by themselves, demonstrate a cause and effect relationship. The interpretation of these findings is also crucially dependent on well-known limitations, including possible sources of bias and confounding, together with the ever-present difficulty in obtaining reliable and accurate population exposure data.

Direct measurements of exposure from incinerators are seldom made and often the distance from the incinerator site is used as a proxy for exposure, a technique that can be very unreliable. Many studies use concentric circles to identify "at risk" populations, a technique that does not take into account the influence of meteorological conditions or process characteristics (e.g. stack height, efflux velocity, plume temperature). Furthermore, the zones of influence used, up to 7.5 km from the site, introduce considerable possibilities for confounding co-exposures to emissions from other industries, past and present.

Another problem is that most studies are, by their very nature, *post hoc* (i.e. after the event) since they were prompted by complaints of apparent "clusters" of ill-health in areas around incinerators. As a result unintentional bias (such as in the reporting of health outcomes) can be built into the study, which can give spurious results. In addition, many studies analyse only a small number of cases which reduces the statistical power to detect an association between exposure and ill-health.

As noted above, the majority of the studies, and any associated environmental data, originate from incineration facilities whose emission profile was significantly different from today's modern incinerators. Up until the mid-1990s, incinerators in the UK were fitted with emission controls of lower efficiency and therefore emitted much larger amounts of air pollutants than is now the case. Currently operational incinerator plants have to meet much stricter controls on emissions and are significantly cleaner (as discussed further in Chapter 2). Where available, we have included details of the period of operation of the incinerator(s) being studied. Some of the studies reviewed were based upon non-MSW incinerators (e.g. sewage sludge or industrial hazardous waste). Since exposure concentrations were not measured, and emissions are not generally reported, the extent to which such incinerators may be considered directly comparable with MSW incinerators is unknown. In qualitative terms, however, emissions are likely to be broadly similar to MSW incinerators in most cases.

The literature review of the field led to the identification of 23 epidemiological studies which have been evaluated. The more important studies are summarised in Table 3.9. Each study was assessed using a critical appraisal according to study type, sample size, exposure definition and measurement, outcome and control for confounding factors. The epidemiological studies which were evaluated are denoted by * in the reference list (Appendix 1). In addition, four review papers were considered (those references are denoted by # in the reference list). Most weight was given to those studies that took into account potential confounding factors, had a valid means of estimating exposure from the incinerator and had sufficient statistical power to produce results with a small confidence interval.

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	posting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

In 2000, the Department of Health Committee on Carcinogenicity in Food, Consumer Products and the Environment agreed that a number of factors should be considered in deriving conclusions from studies of municipal solid waste incinerators (*Committee on Carcinogenicity, 2000*):

- 1. accuracy of health statistics;
- 2. accuracy of cancer diagnosis;
- 3. potential confounding factors for individual cancers; and
- 4. the variables particular to incineration, such as type of waste burnt, geographical and meteorological conditions, and controls placed on the emission of pollutants.

These factors form the basis of this literature review in relation to cancer outcomes.

Potential pathways and exposure routes

The general public can be exposed to pollutants associated with incinerator emissions through a number of routes, with inhalation and the food chain of particular importance. For many pollutants including some of the trace metals and potentially carcinogenic organic compounds (such as dioxins and furans), the major route of exposure is through the food chain.

People with pre-existing respiratory and cardiovascular diseases may have their illness exacerbated by acute exposure to air pollutants such as SO_2 , NO_2 and PM_{10} which occur in incinerator emissions. Children and the elderly are also particularly vulnerable to air pollution. Exposure via the food chain will also arise if locally grown or reared produce is important to the diet of local people. Groups such as local allotment owners and farmers may need particular consideration.

Possible health effects associated with the process

As discussed above, studies in the UK have principally focused on the possible effects of living near to the older generation of incinerators where levels of substances emitted from the incinerator would be expected to be greater than for modern incinerators.

Most concern has focused on the effects of exposure to dioxins and furans and polycyclic aromatic hydrocarbons (PAHs), substances that are known or suspected carcinogens. It has been hypothesised that exposure to dioxins and furans (either directly via inhalation or indirectly via the food-chain) may be major causes of cancer in communities around incinerators. Whilst older incinerators were often significant sources of dioxins and furans in the local environment, modern incinerators are significantly cleaner. A recent study around a modern incinerator in Spain found no difference in the levels of exposure (based on analysis of substances in blood samples) in residents living near to the incinerator and those living further away (Gonzalez et al., 2000).

Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	ıl	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Com	posting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

Cancer

Stomach, colorectal and liver cancers

Several epidemiological studies have suggested a possible association between incinerator emissions and stomach, colorectal and liver cancers. In the UK a possible distance-related link with the old generation of incinerators has been reported (Elliott et al., 1996). This large study examined cancer incidence in over 14 million people living near to 72 municipal solid waste incinerators between 1974 and 1986 (England), 1974 and 1984 (Wales), and 1975 and 1987 (Scotland). Age standardised observed/expected (O/E) ratios were calculated for radii up to 7.5 km from each incinerator and five and ten year lag times for the onset of the cancers were assumed. After controlling for social deprivation, a significant decline in risk with distance from the incinerators was found for all cancers combined, and particularly for stomach, colorectal and liver cancer. However, incomplete control for socio-economic confounders may have been responsible for these results (Elliott et al., 1996) and once the authors took into account a number of post hoc estimates, such as examination of the data before the incinerators were built and estimation of the likely impact of ethnicity, only liver cancer showed a significant association with distance from the incinerators. In this case, it is likely that misclassification of secondary tumours as primary liver cancer may have caused or contributed to the result.

Given the uncertainties that surround this reported excess of liver cancers, particularly the possibility that misclassification of primary tumours contributed to the outcome, the data were re-examined (**Elliott et al., 2000**). Although this study could not completely discount the possibility of an association between residential proximity to municipal solid waste incinerators and incidence of liver cancer, confounding from deprivation (primary liver cancers in Great Britain are strongly related to deprivation) appears to be the most likely explanation for the excess. Even if such a link could be established, this excess relates to historical exposure patterns around older incinerators and not current or future incinerators. In both of these studies, direct measurements of the pollutants were not available and the distance from the incinerator site was used as a proxy for exposure, which is far from ideal.

The waste solvent and oil incinerator at Charnock Richard, Lancashire, which operated between 1972 and 1980, has been much studied. **Gatrell and Lovett (1992)** reported a possible excess of cancer of the stomach and larynx cancer after examining cancer registrations between 1974 and 1983. However, little weight can be given to this study as it did not adequately control for any important confounders. The authors themselves cautioned against making a causal link between cancer incidence and residence near the incinerator. It is also of limited relevance to combustion of municipal waste.

Cancer of the larynx and lung

The possibility of a cancer cluster, particularly of cancer of the larynx, near to the Charnock Richard incinerator prompted a more detailed study which included the other nine UK incinerators licensed to burn waste solvents and oils (**Elliott et al.**, **1992**). Cancer registration data were used to identify cases of cancer of the larynx and lung within 3 km of the sites, and between 3 and 10 km. Expected values were calculated using regionally adjusted national cancer rates to enable age standardised ratios of observed to expected rates to be derived. Cancer of the lung was also included in the study because it shares many of the same epidemiological characteristics as cancer of the larynx, including a strong social class gradient, a



Introduction	Review of Informat on Emissions	tion	Review of Epidemiological Research	I	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Mate	erials Recycling Facilities	Com	posting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

strong association with cigarette smoking and similar occupational risks. However, despite initial reports of a cluster, neither cancer of the larynx nor of the lung showed a statistically significant relationship with distance from the site once socioeconomic status was taken into consideration. Correction for confounding by socioeconomic status was carried out by stratifying the analysis using the Carstairs index of material deprivation, based on the 1981 census. The study concluded that the apparent cluster of cases of cancer of the larynx at Charnock Richard, Lancashire, was unlikely to be due to the incinerator.

A small area study of mortality among residents of Malagrotta, a suburb of Rome, Italy found no association between proximity to industrial sites and mortality from a range of cancers including laryngeal cancer for the period 1987 to 1993 (**Michelozzi et al., 1998**). This area contained a number of industrial point sources, including a waste incinerator that closed in 1985 because of a failure to comply with pollution control standards. Despite no evidence linking mortality from laryngeal cancer or any other cancer with specific sources in the area, there was a marked decline in mortality from laryngeal cancer with distance from the industrial sites. However, the actual dispersion of pollution from these sites was not evaluated and no direct link with the incinerator can be made.

Using a case-control study, **Biggeri et al. (1996)** reported that lung cancer was significantly related to proximity to a waste incinerator and to the city centre in the city of Trieste, Italy after adjusting for confounding factors such as smoking. The study used distance from the source as a measure of exposure and consequently it is difficult to isolate emissions from the incinerator from other sources of air pollution in the area, which included a shipyard and iron foundry. The risk of lung cancer was found to increase with increasing levels of deposition of airborne particulate matter, which greatly exceeded those normally encountered in the UK. The authors drew attention to a number of weaknesses in the study including possible confounding due to other unmeasured exposures, selection bias and misclassification bias due to change in residence.

Childhood cancers

Several studies by Knox have examined a possible association between childhood cancers and industrial emissions including those from incinerators (Knox and Gilman, 1996; 1998; Knox 2000). These studies employ spatial analysis of postcodes of those diagnosed with childhood cancer but limitations with the methodologies used mean that the results of these studies are far from certain. No direct measure of exposure is included in the analysis, with exposure estimates being entirely reliant on using distance from the source as a proxy for exposure. The standardisation technique employed in the earlier studies (Knox and Gilman, 1995; 1998) does not attempt to account for the potential effect of deprivation, which would be a major potential confounding factor. Both of the early studies have been heavily criticised on the basis of lack of proper control for population density and the extreme implausibility of some of the findings, which tentatively linked childhood cancer with a wide range of combustion sources including major highways, but only at considerable distances from the road, at which no elevation in pollutant concentrations from on-road emissions would have occurred.

The most recent study by **Knox (2000)** differs in that it is based upon an analysis of the birth and death addresses of children diagnosed with cancer. This showed a greater incidence of cancer in children born close to incinerators and moving away than in those who moved closer to an incinerator. As its basis, the study assumes that migration of children who subsequently develop cancer should be essentially random. A comparison was made with non-combustion industrial markers including cathedrals, mail order firms and biscuit makers; activities that are not necessarily

Introduction	Review of Informat on Emissions	of Information Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

located in the same sort of areas as incinerators. It is not clear whether hospital incinerators are included in the analysis, since their results are not presented separately. The effect observed by Knox was due to just ten of the incineration plants studied, all but one of which had been open since before 1955 and therefore may have produced substantial historical contamination. All were sited close to other potential sources of environmental hazards. It cannot therefore be taken as representing more modern combustion plants. The work is also liable to criticism on the grounds that there is no information provided on the net migration of total population inwards or outwards from the vicinity of such plants and therefore, again, no control for temporal changes in population densities. The overall conclusion drawn by authors of the study was that the inferred increased probability of childhood cancer stems from residence near to large-scale combustion processes as a whole, of which incinerators are one component.

Soft-tissue sarcomas and non-Hodgkin's lymphomas

Viel et al. (2000) examined the spatial distribution of soft-tissue sarcomas and non-Hodgkin's lymphomas around a French municipal waste incinerator near Besançon, south-west France, with "high" dioxin emissions from 1980 to 1995. The study found localised case clusters of soft-tissue sarcoma and non-Hodgkin's lymphoma in the vicinity of a municipal solid waste incinerator, which were more pronounced at the end of the study period. Again, caution is advised before attributing these clusters to emissions from the incinerator, since as the study did not take into consideration socio-economic status as a contributing factor and there were other uncertainties, due to low spatial resolution of clusters. Furthermore these findings are not consistent with the much more detailed epidemiological study by Elliott and colleagues in the UK, which did not find any association between soft-tissue sarcoma and non-Hodgkin's lymphomas and distance from municipal solid waste incinerators (**Elliott et al., 1996**).

A subsequent paper (Floret et al., 2003) reports a study of greater sophistication into the non-Hodgkin lymphoma cluster (Viel et al., 2000) using a case control study design. A dispersion modelling study of dioxins was conducted and used to assign population to three exposure zones. The study found a significant excess of disease only in the high exposure zone although this finding is surprising given that the predicted dioxin concentrations were far below the usual urban and rural background for dioxins. No measured exposure data were available. The authors reported that no other relevant industrial plant operated within the area and hence, if the effect is causal it would most likely relate to emissions from the incinerator, although given the level of predicted concentrations, it seems unlikely that dioxins were responsible. However, whilst in many ways this study was one of the best designed available, the authors were unable to firmly exclude the possibility that residual confounding affected the reported odds ratios. Levels of dioxin emissions from the incinerator when measured in 1997 were found to be 163 times the normal maximum emission limit for dioxins from incineration.

Conclusion

Despite reports of cancer clusters, no consistent or convincing evidence of a link between cancer and incineration has been published. In the UK, the large epidemiological studies by Elliott and colleagues of the Small Area Health Statistics Unit (SAHSU) examined an aggregate population of 14 million people living with 7.5 km of 72 municipal solid waste incinerators. This included essentially all incineration plants irrespective of age up to 1987. Despite the consequent inclusion of incinerators with emissions of potential carcinogens much higher than would occur from modern incinerators, both studies were unable to convincingly demonstrate an excess of cancers once socio-economic confounding was taken



Introduction	Review of Informat on Emissions	of Information Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

into account (Elliott et al., 1992; 1996; 2000). As a result of these, the Department of Health's Committee on Carcinogenicity published a statement in March 2000 evaluating the evidence linking cancer with proximity to municipal solid waste incinerators in the UK (*Committee on Carcinogenicity, 2000*). The Committee specifically examined the results of these studies and concluded that, "any potential risk of cancer due to residency (for periods in excess of ten years) near to municipal solid waste incinerators was exceedingly low and probably not measurable by the most modern techniques". The Committee agreed that the observed excess of all cancers, stomach, lung and colorectal cancers was due to socio-economic confounding and was not associated with emissions from incinerators. The Committee agreed that, at the present time, there was no need for any further epidemiological investigations of cancer incidence near municipal solid waste incinerators.

It has been hypothesised that exposure to dioxins and furans (either directly via inhalation or indirectly via the food-chain) is responsible for some cancers in communities around incinerators. However, epidemiological studies on the older generation of incinerators that emitted significantly greater amounts of dioxins than newer facilities have failed to identify an effect. Given that the emissions of dioxins and furans from modern incinerators are orders of magnitude lower than from older incinerators, it can be said with some confidence that any impacts of dioxin and furan on cancer rates in local people are small or non-existent and unlikely to be quantified through epidemiology.

Respiratory function

Several studies have examined possible adverse respiratory health effects among people living near incinerators. Perhaps the most credible studies are those which have examined the respiratory health of six communities in North Carolina, USA, three of which are exposed to emissions from biomedical, municipal or hazardous waste incinerators (Shy et al., 1995; Lee and Shy, 1999; Mohan et al., 2000; Hu et al., 2001).

The early study by **Shy et al.** (1995) used objective measures of exposure and respiratory function in a study of three incinerators burning clinical, municipal or hazardous liquid wastes. Indicators of air quality (PM_{10} , $PM_{2.5}$, HCl, HNO_3 and SO_2) were monitored in the areas around these facilities and compared to three matched comparison areas over a 35-day period. The study involved a descriptive investigation of >2,500 households (*c*.7,000 individuals). With the exception of self-reported sinus trouble, chronic cough and wheezing in the area around the hazardous waste incinerator, there were no significant differences between the study and control areas. Aggregating the data for the control and incinerator areas failed to show any excess acute or chronic respiratory symptoms in the incinerator areas. However, some statistically significant differences in the prevalence of respiratory symptoms were reported in a sub-set of individuals from each community who had provided lung function data together with behaviour and health diaries, although analysis of nasal lavage samples did not reveal any effect of living in the vicinity of an incinerator.

The study reported no significant difference of particulate air pollution or overall respiratory health in communities residing near to three waste incinerators from 1992 to 1994. The study found no significant difference in the concentration of PM_{10} in the incinerator communities relative to comparison communities and later analysis of lung function could not confirm any relationship between PM_{10} levels in the communities and lung function (**Lee and Shy, 1999**). Despite some statistically significant differences in prevalence of respiratory symptoms being reported, no consistent differences over the different years of the study were found and there



Introduction	Review of Informat on Emissions	formation s Review of Epidemiological Research Materials Recycling		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recy Facilities	Cor	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

were no differences reported between the different types of waste incinerators studied (biomedical, hazardous and municipal). It therefore seems unlikely that incinerator emissions were a cause of respiratory ill health.

There have been two further published studies on the respiratory health of these communities (Mohan et al., 2000; Hu et al., 2001). In one, over 4,200 respondents were interviewed over the telephone about their respiratory health, smoking and other risk factors (Mohan et al., 2000). Respondents were also asked to provide a subjective assessment of the air quality in their area. The study examined differences in symptom prevalence between each of the study communities and their respective control community and a combined control group. Results indicated a higher prevalence of self-reported respiratory symptoms in one community near the hazardous waste incinerator compared with its control community. After controlling for perceived air quality and when compared with the combined control group, only respiratory symptoms of a long duration remained statistically significant and this result should be treated with caution because of the lack of actual exposure data and the use of self-reported symptoms. As a result, the study is of limited value in evaluating the effect of incinerator exposures.

The other paper examined whether chronic pulmonary effects were related to emissions from the three waste incinerators (Hu et al., 2001). A total of 1,018 subjects underwent a spirometric test once a year between 1992 to 1994. The study attempted to assess exposure using three surrogate measures; living in an incinerator community, distance from the incinerator and an incinerator exposure index, which was a function of the distance and direction of the subjects residence to the incinerator, the number of days the subject spent downwind and the average time spent outdoors. Overall, the test results showed no consistent statistically significant association between pulmonary function and exposure. However, two significant associations were identified, with exposure to the hazardous waste incinerator in 1994 and to the municipal waste incinerator in 1993 being linked with poor forced vital capacity. These associations were present when exposure was estimated based on the distance from the incinerator and also from the incinerator exposure index. However, in neither case can a specific link between pulmonary effects and incinerator emissions be proven. No ambient air monitoring was undertaken prior to the annual spirometric tests and the dispersion of emissions was not considered in the exposure estimate. Furthermore, the time-activity patterns which form the basis of the incinerator exposure index were derived from self-administered questionnaires, which may be subject to bias. It is also worth noting that ambient air monitoring after the annual tests demonstrated that emissions from incinerators did not make a significant contribution to particulate air pollution in these communities and that other sources of pollution would be of more importance.

Gray et al. (1994) examined asthma severity and morbidity among children living in two areas of Sydney, Australia containing high temperature sewage sludge burning incinerators. A total of 713 children aged between eight to twelve years were studied in the two regions close to the incinerators, together with a further 626 children in a control region which did not contain a sludge burning incinerator. Respiratory illness was measured by questionnaire, airway hyper-responsiveness and atopy. The study found no statistically significant differences in the prevalence of current asthma (as defined by airway hyper-responsiveness and recent wheeze), atopy, symptom frequency between the control and two study regions. Furthermore, air monitoring data did not demonstrate any major differences in air quality in the study and control areas. The study concluded that releases from high temperature sewage sludge incinerators appeared to have no adverse effect on the prevalence or severity of childhood asthma. A small study on open air wire reclamation incineration in Taiwan reported a higher incidence of pulmonary effects

Introduction	Review of Informat on Emissions	tion Review of Epidemiolog Research	cal	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

in children exposed to pollution from the incinerator when compared with a nonexposed control population (**Hsuie et al., 1991**). Whilst air monitoring confirmed that air pollution was worse in the exposed area, it was unclear whether other industrial sources were present and the study could not confirm whether this pollution was in fact directly related to emissions from the incineration. The study also did not find any significant difference in the prevalence of cough and wheeze, which tended to contradict the findings of the pulmonary measurements. The findings of this study leave open the possibility that emissions from wire reclamation incineration may have caused pulmonary problems in local children. However, the air pollutant concentrations were not reported, and as the incineration was carried out in the open without abatement, its relevance to modern municipal waste incinerators is extremely limited.

To conclude, available studies have typically examined respiratory health around the older generation of incinerators. Most are based upon self-reported symptoms and therefore may be subject to bias. Overall, there is little evidence to suggest that waste incinerators are associated with increased prevalence of respiratory symptoms in the surrounding population. This is consistent with the data from risk assessments, emissions and ambient air monitoring in the vicinity of incinerators which indicate that modern, well-managed waste incinerators will only make a very small contribution to background levels of air pollution (see also Chapter 5). In many cases, air monitoring data do not demonstrate that emissions from the incinerators are a major contributor to ambient air pollution.

Reproductive problems

Dioxins and furans are known to adversely affect the reproductive system, and therefore a number of studies have investigated effects of incinerators on reproductive health.

It has been suggested that populations living near waste incinerators have a higher probability of giving birth to twins, possibly due to exposure to dioxins, furans and polychlorinated biphenyls (PCBs) in incinerator emissions. Lloyd et al. (1988) reported that during 1980-1983, the incidence of twinning in two areas near to a chemical waste incinerator in Scotland were significantly higher at 20 and 16 per 1,000 when compared with rates in control areas of between 3 and 13 per 1,000. The authors hypothesise that the increased incidence of human twinning rates over the study period was consistent with anecdotal evidence that polychlorinated hydrocarbons and PCB-related compounds were burnt regularly in the late 1970s. However, the study does not specifically link twinning in the exposed human and animal populations to exposure to polychlorinated hydrocarbons and it acknowledges that it would "be premature to attribute causality to this association between air pollution from incinerators and twinning". Furthermore, although maternal age was taken into account in the analysis of human births, several other possible confounding factors were not. No social or personal risk characteristics were included in the study and no data relating to possible hereditary causes of twinning were examined. The study also suffered from a lack of direct exposure data. These strength of these findings have been debated in the medical literature with **Jones (1989)** in particular presenting a sound argument questioning the basis of this study and correctly citing the lack of evidence of any increased environmental (soil) pollution around the incinerator. This incinerator provides an extreme case relative to municipal solid waste incineration since its main function was to burn hazardous industrial waste and emission controls were rudimentary by modern standards. Emissions of dioxins and especially PCBs are likely to have been much greater than for a modern facility. A study of twinning in Sweden failed to find any evidence of spatial clustering of twinning rates in areas near refuse incinerators (Rydhstroem, 1998). In this study, twinning rates before and after the

Introduction	Review of Informat on Emissions	ion Review of Epidemiologi Research	cal	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

introduction of incinerators was examined in 14 areas between 1973 and 1990. In the majority of the study areas, the incidence of twinning was not found to increase once the incinerators became operational. Whilst one area with an incineration plant showed a statistically significant increase in incidence of twinning, it was not consistent with the majority of the study areas, and another area containing a similar incinerator reported a statistically significant decline.

It has been suggested that airborne pollutants associated with incinerators, particularly dioxins and furans, may be associated with changes in the sex ratio of births. Current statistics indicate that in the UK the proportion of male births has decreased over the last fifty years. Sex ratio is thought to be affected by a wide range of biological and environmental factors including race, birth order, parental age, parental hormone levels, timing of conception, ovulation induction, environmental pollutants and socio-economic status. Undoubtedly human sex ratios at birth are partially controlled by the hormone levels of both parents at the time of conception and therefore, in theory, a mechanism exists by which disruption of the parental hormonal system may be detected as a change in sex ratios at birth. Such disruption may have been the cause of the apparent change in sex ratio in populations exposed to high levels of dioxin (TCDD) for a relatively short period during and after the industrial accident in Seveso, Italy in 1976 (Mocarelli et al., **1996**). Levels of dioxin contamination caused by the Seveso incident greatly exceed those arising from old incinerators, and especially from modern incinerators, by several orders of magnitude. Human body burdens increased far beyond those that arise from current day exposure to dioxins and furans.

In the UK, **Williams et al. (1992)** reported a significant excess of female births around two chemical waste incinerator plants in Central Scotland (same incinerator and study area as in **Lloyd et al., 1988**). However, confidence in this outcome is weak as the study lacked a direct measure of exposure to environmental pollution and inadequately considers several possible confounding factors. There is also the possibility that some births may have been misclassified with regard to antenatal exposure. As noted above, exposures from this old incineration plant will have greatly exceeded those from a modern municipal waste incinerator.

A recent study has reported an association between renal function, cytogenetic measurements, and sexual development in adolescents exposed to environmental pollution, including pollutants from waste incinerators which had closed because they exceeded emission standards (**Staessen et al., 2001**). However, the views of the authors that pollutant exposures in this study point to possible delayed development are speculative. The potential for adverse effects was determined by the presence of biomarkers of exposure and effect, but no link to clinically significant adverse health outcomes was established. The fact that individuals living in areas historically contaminated by lead smelters and incinerators had higher pollutant exposures is not surprising, although the study did not take full account of the importance of other routes of exposure (e.g. the diet) or full control for all potential confounders. Furthermore, the industries included in the report are out-dated and would be expected to be more polluting than more modern operations.

Evidence has begun to emerge to suggest that congenital malformations may be associated with environmental pollution. Whilst most studies have focussed on hazardous waste landfill sites, there has been speculation that increased rates of congenital malformations are linked with exposure to dioxins and furans. Such views are consistent with the findings of animal studies which have demonstrated that high exposure to dioxin is associated with congenital malformations, including cleft-lip and palate malformations.

Introduction	Review of Informat on Emissions	ion	Review of Epidemiological Research	l	Quantification Consequence Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revie Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materi F	ials Recycling Facilities	Com	posting	Other W Disposal C	'aste Options	Incineration	n	Landfill	Tables

In Sweden a case study examined the incidence of cleft-lip and palate malformations near an incinerator following reports of a cluster (Jansson and Voog, 1989). The authors interviewed the parents of six children born with cleft-lip/palate but found no common explanatory factor other than the possibility of a hereditary link in three of six cases. The fact that the children lived more than 15 km from an incinerator, and the highest pollutants levels were within 1 km indicates that the incinerator is unlikely to have been responsible. The authors also conducted a study of cleft lip and palate registrations in the borough both before and after the start of refuse incineration, but could find no increase in the total incidence of these malformations.

Recently **ten Tusscher et al. (2000)** reported on a possible association between incidence of cleft lip and palate with the open incineration of chemical waste in Zeeburg in the Netherlands. However, this local increased incidence of orofacial clefts was during the years 1961 up to and including 1969 and the site in question closed down in 1973. It therefore bears little relevance to the current practices of incineration of municipal solid waste.

After adjustment for social class, year of birth, birth order and multiple births, increased risks of lethal congenital anomaly among babies of mothers living close to incinerators (including an industrial hazardous waste incinerator and a sludge incinerator) and crematoria in Cumbria, north west England have been reported in recent research by **Dummer et al. (2003)**. The authors found a significantly increased risk of spina bifida and heart defects in relation to the proximity of incinerators, but not of stillbirth or neonatal death. The study involved 244,758 births to mothers living in Cumbria between 1956 and 1993 and it is acknowledged by the researchers that changes in medical practices over time may have affected the results. Another limitation of the study is that no actual pollution levels around each site have been measured and therefore a function of distance from the incinerators was used as a surrogate for exposure. The work also relates to the older generation of incinerators and acknowledges possible influences of other industrial sources of emissions.

3.6.3 Adoption of quantitative health impact functions

As noted in the above review, the published epidemiological studies of the health of communities living in the vicinity of incinerators have failed to establish any convincing links between incinerator emissions and adverse effects on public health; specifically no impact was demonstrated on the incidence of cancer, respiratory health symptoms or reproductive outcomes. Consequently, the epidemiology specific to incinerators gives no basis for developing quantitative health impact functions and no attempt is made to use it in this way.

A more fruitful approach is to examine the specific substances known to be discharged from an incinerator to model resultant environmental concentrations and to use exposure-response coefficients relating to those specific substances to estimate the magnitude of adverse health outcomes. This has been the focus of another recent study (*Environment Agency, 2003I; draft*). In relation to the classical air pollutants (sulphur dioxide, particulate matter, oxides of nitrogen and ozone), exposure-response coefficients are taken from *COMEAP (1998)*. These coefficients were derived from time series epidemiological studies of urban populations exposed to a pollutant mix frequently dominated by road traffic emissions, and therefore their applicability to incinerator emissions is open to question. However, the transferability of the coefficients has been considered by the Department of Health COMEAP Committee, the deliberations of which are outlined below. The detailed study of incineration (*Environment Agency, 2003I; draft*) has also considered emissions of dioxins and furans, not only via direct inhalation exposure but also

Introduction	Review of Informat on Emissions	ion Review of Epidemiologi Research	cal	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

through food chain transfer for individuals living close to the point of maximum atmospheric deposition. The calculated incremental dioxin/furan intakes have been compared with the recommended tolerable daily intake (TDI) for these substances and the outcomes are summarised below.

The report on incineration (*Environment Agency, 2003I; draft*) also considers the solid waste and liquid effluent outputs of incinerators and other combustion plants and the potential for human exposure to result from them. These considerations lead to the view that there is very limited scope for human exposure to pollutants within such discharges.

3.6.4 Adaptation of study of health effects of combustion plant

In a study conducted for the Environment Agency, and led by the University of Birmingham (*Environment Agency, 2003l; draft*), quantitative estimates have been made of the public health impact of emissions from typical incinerators and other forms of major combustion plant. Sections of that report are summarised here in order to explain the methodology used to estimate the effects of incineration emissions upon public health.

Methodology

The study led by the University of Birmingham for the Environment Agency was collaborative with the University of Sheffield Waste Incineration Centre, the Department of Fuel and Energy at the University of Leeds and the Medical Research Council Institute for Environment and Health at the University of Leicester. The Universities of Leeds and of Sheffield provided information and advice on the pollutant emissions from the various combustion processes and the MRC Institute for Environment and Health reviewed the effects of relevant air pollutants on human health. The role of the University of Birmingham, other than coordinating the project, was to calculate human exposures and estimate public health impacts.

The basic methodology included the following stages:

- estimation of typical pollutant mass emissions and related physical parameters such as stack height, stack gas temperature and efflux velocity based upon measurements from in-service incinerators;
- the application of the ADMS 3 state-of-the-art atmospheric dispersion model using typical UK meteorological data to calculate the distribution of both short term and long term average ground-level concentrations of pollutants attributable to emissions from the incinerator;
- since the numbers of people exposed to given concentrations of pollutant are a function of the spatial distribution of population around the source of emissions, following a review of data on population densities, two uniform population densities were adopted to represent the distribution of population around the source of emissions. Those population densities were as follows: 566 people per square kilometre representative of rural/suburban areas, and 3784 people per square kilometre representative of suburban/urban areas (Table 3.10);
- by overlaying the map of pollutant ground-level concentrations upon a map of population, the numbers of people exposed to specific ground-level concentrations were estimated;

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n 🗏	Landfill	Tables

- by application of the COMEAP coefficients listed in Table 3.11 and 3.12 together with the baseline health rate data listed in Table 3.13, it was possible to estimate directly the numbers of deaths brought forward and the numbers of respiratory and cardiovascular hospital admissions attributable to the pollutants for which COMEAP coefficients were available;
- using the ratio of emissions of carcinogenic metals (nickel, chromium and arsenic) and of polycyclic aromatic hydrocarbons (expressed as benzo(a)pyrene) to emissions of PM₁₀ particulate matter, and a ratio of the coefficient for carcinogenesis due to that species to the coefficient for deaths brought forward due to PM₁₀ converted to a common metric (one coefficient is expressed as absolute risk, the other is as a percentage increase in risk). The increment in cancers attributable to exposure to the chemical carcinogens can be estimated as a number of cancers in a lifetime (taken to be 70 years);
- for dioxins and furans the incremental exposure due to inhalation was calculated and found to be an insignificant proportion of typical human intakes of these species in the UK. A more important pathway of exposure to atmospheric emissions is through deposition to terrestrial surfaces and transfer through the food chain. Using a food chain model, transfers of dioxins and furans and dioxin-like PCBs into locally produced foodstuffs were calculated and used to estimate an incremental exposure for those living at the point of maximum ground-level concentration arising from incinerator emissions;
- other pollutants emitted from the incinerator are also considered, but their calculated ground-level concentrations were below those expected to elicit significant effects in local populations and therefore no health outcomes were calculated in respect of them.

Specifically, the criteria used for detailed consideration and modelling in the study were as follows: either (a) the concentration increment arising from the emissions was a significant component of the overall aggregate airborne concentration, or (b) the sum of the aggregate of the incremental concentration due to the emissions and the background concentration amounted to a significant proportion of a concentration considered likely to present risks to human health. A conservative approach was adopted whereby any pollutant considered in any way to give a plausible cause for concern was short-listed as a priority pollutant and toxicological reviews were conducted on those pollutants. The pollutants selected were as follows:

- lead
- cadmium
- mercury
- nickel
- chromium
- arsenic
- thallium
- boron
- dioxins, furans and dioxin-like compounds (i.e. PCBs)



Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

- polycyclic aromatic hydrocarbons (PAH)
- particles
- hydrogen chloride
- sulphur dioxide
- nitrogen dioxide

Upon further more detailed consideration, lead, cadmium, mercury, thallium, boron and hydrogen chloride were eliminated from detailed consideration of health impacts since their predicted concentrations, allowing for addition of background, fell well below health-based air quality standards or represented on a very minor proportion of typical human intakes (for those pollutants for which air quality standards were not available).

Incinerators are recognised as a source of emissions of a number of air pollutants which themselves cause adverse effects on public health. Exposure-response functions are available from the epidemiological literature for a number of these pollutants. Since these were derived in the main from epidemiological studies on whole urban populations, the Committee on the Medical Effects of Air Pollutants (*COMEAP, 2000*) has expressed some reservations over their use in estimating effects of emissions from specific industrial installations which are elaborated on below. However, the thrust of COMEAP's advice is that the level of confidence with which effects can be estimated is reduced due to the uncertainties over applicability, rather than the differences in exposure contexts rendering the coefficients inapplicable in the case of point source emissions. Thus, taking account of this advice, it is quite permissible to estimate adverse health outcomes using coefficients derived from epidemiological studies of urban populations, but the additional uncertainty should be acknowledged.

The Committee on the Medical Effects of Air Pollutants has reviewed the available literature relating the effects of nitrogen dioxide, sulphur dioxide, particulate matter (PM₁₀) and ozone to respiratory hospital admissions and mortality, and has recommended coefficients which are expected to be directly applicable in the UK context (COMEAP, 1998). These coefficients are listed in Table 3.11. More recently, COMEAP (2001) has recommended a coefficient suitable for use in estimating the effects of exposure to particulate matter (PM_{10}) upon cardiovascular hospital admissions. This coefficient is shown in Table 3.12. To date, whilst epidemiological studies are able to provide other such coefficients, none has been considered by COMEAP as suitable for application in quantification of effects on the public. These coefficients relate to the effects of short-term (i.e. 24-hour) variations in pollutant concentrations upon the health of entire populations. In the case of airborne particulate matter, COMEAP has also considered at length and commented on studies conducted in the United States which provide estimations of loss of life expectancy resulting from long-term (i.e. decades) of exposure to airborne particulate matter, in this case expressed as a concentration of PM_{2.5} The relationship considered most likely to be applicable to the UK particles. population by COMEAP (2001) in relation to benefits of pollution reduction was that abatement by 1 μ g m⁻³ of PM_{2.5} would lead to an increase in life expectancy of between 1.5 and 3.5 days per person averaged across the existing UK population. For a birth cohort born in 2000 and followed for their lifetime, the gain in life expectancy for the same reduction is estimated as between 0.5 and 4.5 weeks.

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

Much of the concern over incineration relates to emissions of dioxins and furans. As explained in greater detail below, exposure-response functions are not available with respect to dioxin and furan emissions, rather a Tolerable Daily Intake (TDI) is recommended which should preferably not be exceeded when averaged over a substantial period of time. The Tolerable Daily Intake is designed to be protective of human populations against all adverse health effects of exposure to dioxins including endocrine disruption and cancer effects, and it is appropriate to compare incremental exposures arising from dioxin emissions with the tolerable daily intake.

Certain other of the substances emitted from incinerators are known to be able to cause cancer in occupationally exposed populations. By extrapolation from the results of occupational epidemiology, it is possible to estimate slope factors, i.e. exposure-response functions for attributable cancers resulting from unit exposure to the substance in question. For the trace metals, chromium, nickel and arsenic, and for polycyclic aromatic hydrocarbons using the compound benzo(a)pyrene as a marker for the entire mixture, the World Health Organisation recommends exposure-response functions which are listed in Table 3.14. These functions provide upper limit risk estimates for the induction of cancer resultant on lifetime exposure to these substances. It should be pointed out that in the case of chromium, it is only the chromium(VI) oxidation state which is carcinogenic. Since the identification and quantification of different valence states of chromium in incinerator emissions is very difficult, data are not available on the proportion of chromium in the carcinogenic oxidation state. The pessimistic assumption was therefore adopted that 100% of the chromium emitted is in the form of the chromium(VI) oxidation state and therefore carcinogenic.

Type of area, classified by DETR	Average population density range (people km ⁻²)
Rural	26 – 281
Suburban/rural	337 – 2419
Suburban	1397 – 3221
Suburban/urban	1652 - 5527
Urban	5465 - 9040

 Table 3.10
 UK population densities (Todorovic et al., 2000)

Table 3.11	Exposure	response of	coefficients	(COMEAP,	1998)
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Pollutant	Health Outcome	Exposure-response coefficient	Data pedigree
PM ₁₀	Deaths brought forward (all causes)	+ 0.75% per 10 µg/m ³ (24 hour mean)	Good (12)
	Respiratory hospital admissions	+ 0.80% per 10 µg/m ³ (24 hour mean)	Good (12)
Sulphur dioxide	Deaths brought forward (all causes)	+ 0.6% per 10 μg/m ³ (24 hour mean)	Good (10)
	Respiratory hospital admissions	+ 0.5% per 10 μg/m ³ (24 hour mean)	Good (10)
NO ₂	See note below	See note below	

Notes: For NO2 a coefficient of 0.5% per 10 $\mu g/m^3$ was used to estimate the effect on respiratory hospital admissions in a sensitivity analysis.

Introduction	Review of Informat on Emissions	tion Review of Epidemiologic Research	al	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

 Table 3.12
 PM₁₀ cardiovascular exposure-response coefficient (COMEAP, 2001)

Pollutant	Health Outcome	Dose-response coefficient	Pedigree
PM ₁₀	Cardiovascular hospital admissions	+ 0.80% per 10 µg/m ³ (24 hour mean)	Good (11)

Table 3.13 Baseline health rate data

Description	Criteria	Annual baseline health rate (per 100,000)
Respiratory hospital	Emergency respiratory hospital admissions	846.4
admissions (ma):	(codes ICD10 J00 to J99) (England)	
Cardiovascular hospital	Emergency cardiovascular admissions	690.5
admissions (cvha):	(codes ICD10 I20 to I52) (England)	
Deaths brought forward (dbf):	Deaths excluding external causes (England and Wales)	983.6

References: (Department of Health, 2001), (ONS, 2000a), (ONS, 2000b).

Table 3.14 WHO unit risks for cancer

Pollutant	Individual lifetime risk	Outcome	Units	Pedigree
Arsenic	1.5 × 10 ⁻³	Lung cancer	(µg m ⁻³) ⁻¹	Moderate (8)
Chromium (VI)	4.0 × 10 ⁻²	Lung cancer	(µg m ⁻³) ⁻¹	Moderate (8)
Nickel	3.8×10^{-4}	Lung cancer	(µg m ⁻³) ⁻¹	Moderate (8)
B[a]P	8.7 × 10 ⁻⁵	Lung cancer	(µg m ⁻³) ⁻¹	Moderate (8)
Benzene	6.0×10^{-6}	Leukaemia	(µg m ⁻³) ⁻¹	Moderate (8)
Chloroethene	1.0 × 10 ⁻⁶	Haemangiosarcoma	(µg m ⁻³) ⁻¹	Moderate (8)

From: World Health Organisation (2000)

Critical appraisal

There are a limited number of epidemiological studies on populations around incinerators and the results of these are typically inconsistent and inconclusive. Based on current epidemiological evidence it is difficult to establish causality, particularly once confounding factors such as socio-economic variables, exposure to other emissions, population variables and spatial/temporal issues are taken into account. In reality, most data on the possible health effects of incinerator emissions are derived from risk assessments, which are routinely used to evaluate the potential for both direct and indirect carcinogenic and non-carcinogenic risks from proposed installations. Whilst such assessments can help public health professionals identify chemicals of concern, they cannot give exact estimates of the level of risk at the concentrations actually emitted into the environment. However, a quantitative risk assessment approach has been used in this study, as the only means of quantifying the outcomes of air pollutant exposures. One disadvantage of this approach is that it is unlikely to encompass conditions of abnormal operation. However, the negative results of powerful epidemiological studies such as Elliott et al. (1996, 2000) attest to the fact that such conditions are not having a major impact upon health.

Introduction	Review of Informat on Emissions	ion	Review of Epidemiologica Research	I	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envi Revi Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Mate	rials Recycling Facilities	Corr	posting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

Application of COMEAP methodology to calculate deaths brought forward and hospital admissions

The Department of Health Committee on the Medical Effects of Air Pollutants (COMEAP) have developed a methodology applicable to the results of time series epidemiological studies which allows calculation of the public health impact of exposure to the classical air pollutants in terms of the numbers of "deaths brought forward" and the "number of hospital admissions for respiratory disease brought forward or additional" (COMEAP, 1998). Only acute effects of exposure are Whilst this methodology was applied by COMEAP to the urban quantified. population of Great Britain, it is however, possible to apply it to smaller areas and to calculate incremental impacts from developments such as new incinerators. COMEAP has issued a statement expressing some reservations about this development, but nonetheless recognised that it is broadly acceptable to estimate health impacts in this way, provided certain caveats are taken into account. Those caveats generally relate to the fact that the exposure-response coefficients used in the COMEAP report derive from studies of whole urban populations where the air pollution climate may differ from that around a new industrial installation.

In their report entitled "The Regulatory and Environmental Impact Assessment of the Proposed Waste Incineration Directive" which has become known as WID REIA, the consultants Entec applied a COMEAP type methodology to hypothetical incinerators, deriving health impacts in terms of deaths and hospital admissions per tonne of pollutant emitted (*Entec, undated*). Such results are specific to the precise scenarios in terms of stack height, plume rise, population density etc used by Entec and are therefore not directly applicable to other emissions scenarios. Despite this fact, they have been widely applied to the estimation of deaths and respiratory hospital admissions relating to proposed industrial developments.

In the original Entec report, by far the largest impacts were attributed to nitrogen dioxide through its indirect effect on the production of ozone. In a subsequent erratum, Entec admitted to a large error in their coefficient and recommended an alternative coefficient for nitrogen dioxide. Whilst the level of detail of their calculations presented in the REIA report is extremely limited, it does appear that the COMEAP methodology has been applied incorrectly in respect of all pollutants and therefore the results are in error. Furthermore, the coefficients in the report relating to deaths per tonne of pollutant and hospital admissions per tonne of pollutant appear to be erroneous.

In response to such use and concern that the methodology was being misused, members of COMEAP discussed the applicability of using these time-series coefficients to areas affected by emissions of air pollutants from industrial sources (*COMEAP*, 2000). They agreed that coefficients reported in time-series epidemiological studies linking concentrations of air pollutants and measures of illhealth, could be used to estimate the effects of air pollutants emitted by industrial processes on health of people living in areas affected by such emissions, provided that the uncertainties of this approach are acknowledged. However, whilst it was accepted that such an approach might provide useful estimates of effects on health, COMEAP indicated that the extent of these uncertainties could not, at present, be established. COMEAP identified a number of factors and assumptions that would contribute to the uncertainty of the estimates. These were:

i. In applying the above approach it is assumed that the spatial distribution of concentrations of the air pollutants considered is the same in the area under study as in those areas, usually cities of large towns, in which the studies which generated the coefficients were undertaken.

Introduction	Review of Information on Emissions	Review of Epidemiologica Research	I	Quantificati Consequen Emissions	Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Com	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

- ii. It is assumed that the temporal pattern of pollutant concentrations in the area under study is similar to that in the areas in which the studies which generated the coefficients were undertaken (i.e. urban areas).
- iii. It should be recognised that a difference in the pattern of socio-economic conditions between the areas to be studied and the reference areas could lead to inaccuracy in the predicted level of effects.
- iv. In the same way, a difference in the pattern of personal exposures between the areas to be studied and the reference areas will affect the accuracy of the predictions of effects.

It will be seen from the above that some comparisons of the study areas and the reference areas should ideally be undertaken before making calculations of effects. For instance, it might well be unwise to use coefficients derived from studies in towns undertaken in deprived urban areas to predict effects in comparatively affluent rural areas.

There are also a number of assumptions regarding the toxicological properties of the air pollutants considered under the COMEAP methodology. These include:

- i. Linearity of the relationship between ambient concentrations and effects. This assumption is well supported for acute effects of the classical air pollutants at typical ambient concentrations.
- ii. In the case of particles it is assumed that the toxicity of the ambient aerosol represented by a measure of the mass concentration of a specified fraction of the aerosol, e.g. PM_{10} , in a study area is similar to that in areas where the exposure-response coefficients were derived. Most epidemiological studies have been conducted in urban areas where transport-generated particles make a significant contribution to the ambient aerosol. Application of coefficients from such studies to areas in which transport-generated particles make only a small contribution will include an element of uncertainty.
- iii. The annual average concentration of pollutants is frequently used as a basis for calculations: its use implicitly assumes that the effects of the pollutants are not characterised by a threshold of effect.
- iv. If coefficients for several pollutants are applied and the calculated effects summed, it is assumed that the pollutants act independently and that the coefficients have been derived from studies in which this was tested and found to be supported by the evidence. Adding the effects attributed to particles and ozone is likely to be valid. The case for adding the effects attributed to particles and sulphur dioxide is less strong but probably acceptable. It would clearly be wrong to add the effects attributed to PM₁₀ to those attributed to PM_{2.5}. Whether effects attributed to particles should be added to those that might be attributed to nitrogen dioxide or carbon monoxide seems dubious and this was not done in the COMEAP quantification report (COMEAP, 1998).

Finally, COMEAP recommended that the following be noted.

i. Estimates of effects made as described above exclude the possible effects of long-term exposure to air pollutants. Evidence to show that such effects may well be important has accumulated recently: the interpretation of such evidence is still under consideration.

Introduction	Review of Informat on Emissions	tion Review of Epidemiolog Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

- ii. Coefficients are available for only a small group of air pollutants: no quantitative estimate of effect can be made for other pollutants.
- iii. If estimates of effect are made for very small areas it is likely that only small numbers of for example, deaths or hospital admissions will be generated. It would be unwise to put too much weight on small differences between already small numbers: for example, two extra deaths as compared with one extra death.

Results

The results of the calculations using COMEAP coefficients and World Health Organisation Unit Risk factors for carcinogenesis appear in Tables 3.15 and 3.16. The data have been extracted directly from *Environment Agency (2003l; draft)* and have also been normalised according to the incinerators' capacities in order to produce figures for the health impacts per tonne per waste combusted for the two incinerators considered in the report. These were selected on the basis of stack height, one being close to the median for UK incinerators, the other being one of the lowest stack heights. In the case of the chemical carcinogens, emissions data for the relevant substances were available only for the median stack height case.

Health Outcome	Incinerato (median stack	r A height)	Incinerator B (lower stack height)			
nealth Outcome	Annual effect	Effect per tonne MSW	Annual effect	Effect per tonne MSW		
Deaths brought forward						
- sulphur dioxide	0.0079	6.4 x 10 ⁻⁸	0.0107	1.0 x 10 ⁻⁷		
- PM10	< 0.00005	< 4.1 x 10 ⁻¹⁰	0.0010	1.0 x 10 ⁻⁸		
Respiratory hospital admissions						
- nitrogen dioxide	0.186	1.5 x 10 ⁻⁶	0.4343	4.1 x 10 ⁻⁶		
- sulphur dioxide	0.0057	4.6 x 10 ⁻⁸	0.0077	7.3 x 10 ⁻⁸		
- PM10	< 0.00005	< 4.1 x 10 ⁻¹⁰	0.0010	1.0 x 10 ⁻⁸		
Cardiovascular hospital admissions	- 0.00005	- 4 1 × 10 ⁻¹⁰	0.0008	7.6 × 10 ⁻⁹		
- PINITU	< 0.00005	< 4.1 X 10	0.0008	7.6 X 10		

 Table 3.15
 Quantitative impacts of municipal incinerator emissions – classical pollutants – urban population density (3784 km⁻²)

Table 3.16 Quantitative impacts of municipal incinerator emissions – chemical carcinogens - urban population density (3784 km⁻²)

Pollutant	Cancers/70 yrs	Cancers per tonne MSW
Arsenic	< 0.00005	< 5.8 x 10 ⁻¹²
Chromium (VI)	< 0.00005	< 5.8 x 10 ⁻¹²
Nickel	< 0.00005	< 5.8 x 10 ⁻¹²
Polycyclic Aromatic Hydrocarbons	< 0.00005	< 5.8 x 10 ⁻¹²

Introduction	Review of Informat on Emissions	tion Review of Epidemiolog Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

As explained above, in the case of dioxins and furans, the contribution of incinerator emissions to direct respiratory exposure is a negligible component of the average human intake. However, the substances may make a larger contribution to human exposure via the food chain and calculations were conducted to estimating incorporation into locally grown foodstuffs for a person living at the point of maximum ground-level concentration with respect to these substances. The results were expressed as a percentage increment of existing modelled background exposure, and in the case of municipal waste incinerators using actual emissions led to estimates of 0.66% and 0.80% of modelled background exposure. Thus, even for people living close to the point of maximum ground-level concentration and consuming a significant proportion of locally grown foodstuffs, the contribution of incinerator emissions to their intake of dioxins and furans is less than 1%. Current UK adult intakes of dioxins/furans are close to the tolerable daily intake (TDI) recommended by the Committee on Toxicity of Chemicals in Food, Consumer Products and the Environment of 2 picograms per kg body weight per day. Adopting the approach recently applied in regard to exposures from contaminated land (Environment Agency, 2003), this increment is well below a level that would be of concern. The TDI for dioxins/furans is based on reproductive effects in rodents but because of the non-genotoxic carcinogenic nature of dioxins/furans, is expected to be protective against all adverse effects in human populations of exposure to dioxins, including cancer and adverse reproductive endpoints.

The Environment Agency (2003l; draft) report also examined the following issues:

Disposal of solid waste residues from incinerators. It was concluded that if proper procedures are followed there should be negligible public exposure resulting from disposal of solid incinerator residues, including furnace bottom ash and air pollution control residues and that therefore the public health impact will be effectively zero.

Discharge of contaminated cooling waters from incinerators. The scenarios resulting in possible human exposure due to discharges of contaminated waters either into municipal sewage systems or surface waters were examined, and provided normal controls are maintained on sludge disposal and drinking water quality, the public health impact of contaminated water disposal in this way should also be negligible.

The application of dose-response functions to other waste management facilities is described in Chapter 4.

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantificati Consequen Emissions	Quantification of Health Consequences of Emissions		cation of the mental uences of ns	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

3.7 Landfill

3.7.1 Introduction

As described in Chapter 1, the practice of landfill involves the use or creation of contained void spaces. These are normally in the form of cells which can be lined, then filled with waste materials which are progressively compressed and enclosed with further soil, and eventually with a permanent cap. Since much of the waste is not processed prior to disposal in a landfill, biodegradable materials subsequently decay releasing landfill gas. Landfill gas comprises mainly methane and carbon dioxide, and is increasingly collected for combustion and energy conversion.

Detailed chemical analysis of landfill gas shows the presence of potentially toxic components to which of adjacent populations could be exposed due to incomplete collection of gas. Landfills are susceptible to the ingress of water principally from rainfall. Modern landfills are lined with a comprehensive low permeability system which limits seepage of leachate to surrounding soils to a level assessed to be acceptable, and capped when full. For older landfills, however, greater movement of leachate is a potential pathway for human exposure, as set out in Chapter 2.

Concern over the health effects of landfill stems in the main from historic poorly regulated industrial waste sites from which contamination of the local environment is in some cases well documented. As noted in the general introduction, however, municipal solid waste can include some hazardous materials from domestic sources. Both the disposal of hazardous materials and their production by biodegradation processes can lead to the potential for environmental releases of hazardous materials from municipal solid waste.

3.7.2 Review of health effects studies

The majority of published research on the human health effects of landfill relates to landfill sites which accepted either hazardous waste or co-disposal of municipal and hazardous wastes. Such sites are outside of the scope of this study and therefore many superficially relevant studies have been deliberately excluded.

Redfearn and Roberts (2002) have presented a detailed review of the available epidemiological literature on landfill and health. They separate the available epidemiological studies into four categories as follows:

- Single site studies of waste sites including hazardous waste sites, illegal landfills or "in-house" landfills within the curtilage of industry;
- Multi-site studies of sites including hazardous waste sites, illegal landfills or "inhouse" landfills within the curtilage of industry;
- Single site epidemiological studies of potential health effects associated with landfill including some sites accepting hazardous waste;
- Multi-site epidemiological studies of potential health effects associated with waste disposal sites, some accepting hazardous waste.

Redfearn and Roberts (2002), discounted the first two groups of studies as concerning sites which did not in any way parallel current UK landfill practice, and which were therefore not useful in interpretation of effects. The papers in the latter two categories are summarised in Tables 3.17 and 3.18. **Redfearn and Roberts**



Introduction	Review of Informat on Emissions	ion	Review of Epidemiologica Research	I	Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Mate	rials Recycling Facilities	Corr	posting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

(2002) went on to analyse the outcomes of the various studies in terms of demonstration of excess risks. The summary table which they presented appears as Table 3.19. This categorises studies according to health outcome and whether the study indicated an excess risk for those residing in the vicinity of a landfill for that health outcome and those indicating no excess risk. Those reported as demonstrating excess risk showed a significant positive association between a health outcome and proximity to a landfill site. Those indicated as showing no excess risk did not show a statistically significant association, although the reason could be lack of statistical power to demonstrate such an association, in which case the lack of a demonstrated excess risk should not be taken necessarily as an absence of risk. The majority of the adverse health outcomes studied come under the categories of birth defects and other pregnancy outcomes, and cancers. The balance between studies with and without a positive finding appears more strongly in favour of outcomes with an excess risk in the case of birth defects as opposed to cancer. Thus, whilst Table 3.19 is useful in illustrating the diversity of results from the various studies, it should not be used to infer adverse effects caused by landfill. Rather, it is necessary to examine individual studies to draw conclusions in this matter. It must also be remembered (see Section 3.2.1) that statistically significant associations can occur purely by chance especially in studies where a large number of possible relationships are examined.

Many of the landfill sites listed in Table 3.17 are notorious toxic waste sites which are known to have caused problems of one kind or another. Sites such as the Nant-y-Gwyddon landfill in Wales which initially gave concern of malodorous emissions has been the subject of a number of studies and adverse health effects remain controversial. Many of the other sites were in North America and had a long history of poorly controlled disposal of hazardous wastes. Table 3.18 includes some of the more modern and comprehensive multi-site studies including the pan-European EUROHAZCON study (Dolk et al., 1998; Vrijheid et al., 2002). With the exception of the Elliott et al. (2001a,b) studies, the research addressed hazardous waste sites explicitly and is not of direct and immediate relevance to this study of municipal waste disposal.

The key study in the UK context is that by the Small Area Health Statistics Unit (**Elliott et al., 2001**). This was a study of adverse birth outcomes in populations living near landfill sites where the "exposed" population was defined as living within 2 km of one of 9565 landfill sites operational at some time between 1982 and 1997, when compared with those living further away. All of the landfill sites were located in Great Britain and the study examined 124,597 congenital anomalies (including terminations) amongst over 8.2 million live births and 43,471 stillbirths.

The sole criterion used by Elliott et al. (2001) for judging exposure to the landfill activity was proximity of residence. For 70% of landfill sites, distances were measured from the site centroid whilst for the remainder the location of the site gateway at the time of reporting was used. A 2 km zone was constructed around each landfill site corresponding to an assumed likely limit of dispersion for landfill emissions. Persons with residential postcodes within the 2 km zone were classified as within the exposed population, whilst people living more than 2 km from all known landfill sites during the study period comprised the reference population. Fifty-five percent of the national population resided within the 2 km zones around the 9565 landfill sites operational between 1982 and 1987, which comprised 774 sites for hazardous waste, 7803 sites for non-hazardous waste and 988 sites which handled unknown wastes. Congenital anomalies which were examined included neural tube defects, cardiovascular defects, abdominal wall defects, hypospadias and epispadias, surgical correction of hypospadias and epispadias and surgical correction of pastroschisis and exomphalos. The instances of low and very low

Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

birth weights defined as less than 2500 g and less than 1500 g respectively were also examined.

Risks for the population within 2 km of landfill relative to the reference population were calculated by indirect standardisation assuming a common relative risk for all landfill sites. The regression function included year of birth, administrative region, sex, (for birth weight and still births) and deprivation. The latter was based on the national distribution of the Carstairs deprivation index based on 1991 census statistics at enumeration district level. The results for risks of congenital anomalies, stillbirths and low birth weights during operation or after closure of a landfill site combining all waste types appear in Table 3.20. After the important adjustment for deprivation, there remains a small but nonetheless statistically significant excess relative risk for those living within 2 km of a landfill site for all congenital anomalies, neural tube defects, hypospadias and epispadias, abdominal wall defects, surgical correction of gastroschisis and exomphalos, low birth weight.

The analysis was also carried out separately for sites handling special (i.e. hazardous) waste and non-special waste as well as for sites that opened during the study period, relative risks before opening and during operation or after closure. The results appear in Table 3.21. The authors comment that sites listed as handling special (i.e. hazardous waste) due to the UK practice of co-disposal of special and non-special wastes may in fact only handle small volumes of hazardous waste; they are likely to be subject to stricter management and design standards than other UK sites therefore minimising pollutant releases and exposure of the local population. On the other hand, the authors raise the possibility that hazardous waste may have been disposed of unreported in non-special waste sites. Given the strict regulatory regime in place, this appears unlikely to have occurred in practice in recent years. The results indicate that for the statistically significant associations of birth outcomes with residence within 2 km of a landfill site, the relative risk appears to be greater for special waste than non-special waste sites. Those birth outcomes which show an excess of disease for non-special waste sites are all congenital anomalies combined, neural tube defects, hypospadias and epispadias, surgical correction of gastroschisis and exomphalos, low birth weight and very low birth weight. For the latter two outcomes the relative risk is marginally higher for non-special waste sites than for special waste sites.

When risks associated with sites that opened during the study period (irrespective of waste type) were compared over the periods before opening with those during operation or after closure, rather few of the estimated relative risks were significant. Whilst relative risks were higher for some birth outcomes during operation or after closure of the site, for certain birth outcomes, most notably abdominal wall defects, the relative risk before opening of the site was greater than during operation or after closure. Whilst stillbirths, low birth weights and very low birth weights were all significantly associated with residence within 2 km of a landfill site during operation or after closure, prior to opening none was significantly associated. The authors comment that this latter kind of analysis involving rates of disease both before and after the opening of landfill sites being restricted to one set of areas is less subject to confounding by socio-demographic factors than comparisons between different areas, although confounding by temporal trends is possible.

Commenting on the paper by Elliott et al. (2001), McNamee and Dolk (2001) drew attention to the fact that small errors in adjusting for confounding, for example, by socio-economic class could increase or decrease relative risk for landfill versus reference areas quite appreciably. They also questioned whether residence within 2 km of a landfill was the best measure of exposure and pointed out various reasons why misclassification of exposure might have occurred. For example,

Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

because the study was based on residence at pregnancy outcome, misclassification would occur if women moved home between the critical period of early pregnancy and the end of pregnancy.

Whilst there are weaknesses in the Small Area Health Statistics Unit study (**Elliott** *et al.*, **2001**), it is undoubtedly the strongest piece of epidemiological research carried out in the UK and probably internationally on the issue of risks of congenital anomalies in relation to landfill. The small positive association found between certain adverse birth outcomes and residence in proximity to a landfill cannot be stated with certainty to be causal, but provide the best currently available estimate of relative risk.

Although not included in the main published paper, the study also examined a number of cancer outcomes, specifically childhood and adult leukaemias, hepatobiliary cancers and cancers of bladder and brain. After controlling for socioeconomic status, no excess risk for those living within 2 km of a landfill site was found for each of the cancer types studied (**Jarup et al., 2002**). This result must be viewed with less confidence than those relating to congenital malformations because of the likely latency period in developing a cancer. SAHSU used a lag period of one year for childhood leukaemia and five years for the other cancer to increase the number of years of data available for analysis and to reduce the potential of dilution by migration. If, however, the latency period is longer, this index of potential exposure may be inappropriate leading to dilution of any potential effect (*Committee on Toxicity, 2001*).

3.7.3 Applicable exposure-response functions from literature review

The available information regarding studies of landfill and the direct effects on the health of local populations has been reviewed.

Elliott et al. (2001) provides quantitative estimates of excess risk of congenital anomalies, stillbirths and low and very low birth weights in populations living within 2 km of a landfill site. Based upon Table 7 of **Elliott et al. (2001)**, and the column referring to non-special waste sites, the attributable increments in adverse health outcomes for those living with 2 km of a landfill site are as shown in Table 3.22.

Whilst the estimates of excess disease attributable to municipal waste landfills set out in Table 3.22 are the best currently available, they should be treated with some caution. The study was not able to state whether the observed increment was due to exposure to emissions from the landfill, or to some other cause or combination of causes. The relatively small scale of incremental health risks means that we are less confident that the reported effects are in fact caused by a particular cause or combination of causes such as the landfill sites studied. As discussed in the review of this paper, the results are sensitive to possible misclassifications of socioeconomic status, and it is possible that the outcomes are the result of residual confounding rather than a true reflection of an excess of disease attributable to the landfill.

Furthermore, doubt is attached in some cases to the exact nature of waste that was being disposed: in those cases, there is some question as to whether the sites categorised as "non-special waste" may have received some hazardous wastes as part of, or in addition to municipal refuse. If this has occurred, this may have affected emissions from these sites. Finally, the study investigated sites which opened during the period covered by the study, comparing the rates of disease before and after the sites opened. It was found that some of the outcomes considered were at a lower rate after the site opened than before it opened. This



Introduction	Introduction Review of Informa on Emissions		Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Composting		Other Waste Disposal Options		Incineration		Landfill	Tables

indicates that factors other than the landfill sites were at least contributing to the observed increases.

Table 3.22	Increments in adverse health outcomes for populations within 2 km of a landfill
	site

Outcome	Observed increment (99% confidence interval)
Neural tube defects	6% (1-12%)
Cardiovascular defects – no excess	No excess
Hypospadias and epispadias	7% (4-11%)
Abdominal wall defects	7% (-1-12%)
Gastroschisis and exomphalos (surgical corrections) ¹	18% (3-34%)
Stillbirths – no excess	No excess
Low birth weight	6% (5.2%-6.2%)
Very low birth weight	4% (3-6%)

Note 1: Derived from Elliott et al. (2001)

Note 2: Surgical correction of gastroschisis and exomphalos was included in the study of Elliot et al. as a cross-check on the data for abdominal wall defects. The cases in this category are also included in the wider category of abdominal wall defects.

Introduction	Review of Informat on Emissions	tion	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Glossary	Introduction	Mater	rials Recycling Facilities	Corr	posting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables	

3.8 Tables

164

Introduction	Review of Informat on Emissions	tion Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions	
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal (/aste Options	Incineration	ı	Landfill	Tables	

Table 3.1 Pollutants arising from sorting of unseparated waste (from Gladding, 2002)

Author	Country	Facility-type	Contaminants & Concentrations
Diaz e <i>t al</i> (1976)	USA	Resource Recovery Facility	 Aluminium/cadmium & asbestos were undetectable, iron 374.1 µg/g and lead 235 µg/g (in particles sampled). Bacteria not over 0.36 x 103 cfu/ft air. In the waste Faecal Coliforms reached 10⁶, Faecal Streptococci 10⁶ cfu/m3.
Constable <i>et al</i> (1979)	UK	Mixed Waste Sorting/RDF plant	 Penicillium to 5.8 x 10⁴ cfu/m3, and bacteria high (not specified). Few Aspergillus, Cladosporium, Actinomycetes.
Mozzon <i>et al</i> (1987)	USA	RDF plant	 Lead up to 2.1 mg/m³ at the precipitator cleaner, though more often at 0.003 mg/m³. Cadmium peaking at 0.32 mg/m³ in the same area, and in other parts 0.003 mg/m3. Asbestos and PCBs were not found in any samples.
Crook <i>et al</i> (1987)	UK	RDF plant	 Bacteria up to 10⁶, fungi 10⁷ and actinomycetes 10⁵ cfu/m³. Gram-negatives and pathogenic microorganisms caused concern.
Malmros <i>et al</i> (1988) Sigsgaard <i>et al</i> (1990) Malmros <i>et al</i> (1992) Malmros <i>et al</i> (1994)	Denmark	Mixed waste sorting to produce RDF pellets	 Viable counts showed microorganisms to 20,000 cfu/m³ in reception and during manual sorting. Endotoxins were highest at the RDF press, 0.99 µg/m³. Rebuilding of the plant reduced these to 8,400 cfu/m³ as the highest reading at the magnet and 0.11 µg/m³ during sorting respectively.
Rahkonen (1992)	Finland	Mixed Waste Sorting	 Lead to 0.26 μg/m³ & cadmium to 0.09 μg/m³ at the sorting belt. Bacteria to 1.4 x 10⁴, fungi to 2.5 x 10⁴ cfu/m³. Total dust to 38 mg/m³, Endotoxin to 30 ng/m³.
Pfirrmann and vanden Bossche (1994)	Germany	Waste Removal Plant	 Viral infectivity in 12 of 36 samples, belonging to the family Picornaviridae.
Marchand <i>et al</i> (1995)	Canada	Mixed Waste Sorting	◆ Bacteria to 5.2 x 10 ⁵ , Gram-negatives to 7.9 x 10 ³ , fungi to 7.2 x 10 ³ cfu/m ³ .
Jager <i>et al</i> (1995)	Germany	'Garbage' Sorting	◆ Bacteria to 1.4 x 10 ⁴ , Gram-negatives to 7.2 x 10 ³ , fungi to 8.4 x 10 ⁴ cfu/m ³ .
Streib <i>et al</i> (1996)	Germany	Mixed Waste Sorting	 Airborne natural and artificial fibres found occasionally. Cadmium, mercury and nickle in the range of their natural concentrations in urban areas, lead found in excess. Microorganisms 6.9 x 10⁵ cfu/m³, fungi 6.6 x 10⁴ cfu/m³, with 90% below 7 microns. Exposure limit for dust of 6 mg/m³ exceeded for short periods.
Van Tongeren <i>et al</i> (1997)	N.L.	Resource Recovery Facility	 Dusts up to 14.3 mg/m³ during manual separation of waste, endotoxin from 32 ng/m³ to 131 ng/m³. Microorganisms to 10⁶ cfu/m³ during tipping.
Mahar <i>et al</i> (1999)	USA	Two RDF plants	• Total dust geometric mean 0.50mg/m ³ , endotoxin 2.9 ng/m ³ , 6.8 x 10 ⁵ cells/m ³ total microorganisms.



Introduction	Review of Informat on Emissions	Review of Information on Emissions Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	ı	Landfill		Tables

 Table 3.2
 Pollutants arising from sorting of separated waste (from Gladding, 2002)

Author	Country	Facility-type	Contaminants & Concentrations
Nersting <i>et al</i> (1991)	Denmark	Source segregated waste	◆ Bioaerosols from 6 x 10 ² to 4.7 x 10 ⁴ cfu/m ³ . One sorting hall with no ventilation/temporary shielding had fungi levels of 1.4 x 105 cfu/m ³ and 5.4 x 10 ⁵ cfu/m ³ Gram-negative bacteria were found when large quantities of waste were sorted. Endotoxins up to 14.41 ng/m ³ . Dust concentrations less than 5 mg/m ³ .
Sigsgaard (1993)	Denmark	Source segregated waste compared to paper and composting operatives	 Fungi levels around 1.4 x 104 cfu/m³ and bacteria between 5 x 10³- 10⁵ cfu/m³ in waste handling plants. Significantly higher endotoxin concentration was found in waste handling plants compared to paper sorting plants.
IEERR (1995)	USA	Six MRFs	 Environmental measurements Pb 0.07 µg/m³, Hg 0.006 µg/m³. Occupational measurement silica 0.12 mg/m3 & metals all below 0.01 µg/m³. Total dust up to 2.50 mg/m³, respirable 0.57 mg/m³. Peaks of micro-organisms at different sites seen in Table 3.5. Suspended particulates up to 122.75 µg/m³.
Sigsgaard <i>et al</i> (1996)	Denmark	Source segregated waste compared to paper and composting	 Lead up to 3.9 μg/l, mercury 2.3 μg/l in the blood. Cadmium up to 3.6 μg/l in blood of waste workers compared with 1.7 μg/l in controls. Total dust highest in waste handling plants, at 0.74 mg/m³. Waste handling and compost plants showed the highest viable counts of bacteria and fungi up to 83 x 10⁴ cfu/m³.
Gladding and Coggins (1997)	UK	Source segregated MRFs	◆ Bacteria and fungi to 2.5 x 10 ⁵ cfu/m ³ with total dust levels to 18 mg/m ³ .
Kivranta <i>et al</i> (1999)	Finland	Source segregated sorting	 Viable fungi, bacteria and Gram-negative bacteria to 10⁵ cfu/m³, VOCs peaking at 3000 μg/m³ considered to be the limit for discomfort.
Lavoie and Guertin (2001)	Canada	Segregated materials	 Bacteria to 2.1 x 10⁴ cfu/m³, Gram-negative bacteria to 3.2 x 10³ cfu/m³, fungi to 1.4 x 10⁴ cfu/m³. CO₂, CO, NO and NO₂ not measured in significant amounts. EMFs low, noise exceeded 90 dB(A) in one plant. Ergonomics a possible risk factor for MRF workers.



Introduction	Review of Informat on Emissions	tion Review of Epidemiolog Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions	
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other Waste Disposal Options		Incineration		Landfill	Tables	

Table 3.3	Exposure vs. number of workers suffering named symptom (in %) in a materials
	recycling plant (from Gladding, 2002)

Symptom/Exposure	Higher Exp.	Middle Exp.	Lower Exp.	Linear	Chi
Total Dust					
Itchy Red Skin	4.8	20.5	7.7	0.648	0.026*
Skin Rash	2.4	10.3	0	0.691	0.043*
Irritated nose/sneezing	66.7	50.0	46.2	0.062	0.125
Diarrhoea	45.2	44.9	20.5	0.026*	0.024*
Flu symptoms	21.4	42.3	13.2	0.493	0.002*
Endotoxin					
Cough with phlegm	33.3	20.4	17.1	0.048*	0.114
Dry cough	37.3	16.3	40.0	0.760	0.022*
Stuffy nose	64.0	61.2	45.7	0.090	0.180
Hoarse/parched throat	33.3	12.2	20.0	0.048*	0.022*
Glucan					
Cough with phlegm	33.3	28.6	16.7	0.039*	0.107
Hoarse/parched throat	31.6	28.6	13.3	0.021*	0.049*
Chest tightness	15.8	11.9	5.0	0.059	0.161
Stuffy nose	66.7	57.1	53.3	0.145	0.326
Irritated nose/sneezing	47.4	66.7	50.0	0.796	0.130

*Indicates significant association between exposure and health (p=0.05)

167

Introduction	Review of Informat on Emissions	ion Revie Epide Resea	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions	
Glossary	Introduction	Materials Re Facilitie	Recycling ities	Com	nposting Other W Disposal C		'aste Options	Incineration	n Landfill		Tal	oles

Table 3.4:Adjusted odds ratio for symptoms vs exposure for workers in a recycling plant
(from Gladding, 2002)

Symptom	Low vs. Medium	Low vs. High
Total Dust		
Irritated nose/ sneeze	1.0819 (0.4848-2.4140)	2.6869 (1.0476-6.8914)*
Diarrhoea	3.4162 (1.3761-8.4807)*	3.5559 (1.2945-9.7676)*
Flu Symptoms	3.6438 (1.3496-9.8383)*	1.3651 (0.4278-4.3559)
Endotoxin		
Cough with phlegm	1.2758 (0.4112-3.9586)	2.7082 (0.9724-7.5424)
Stuffy Nose	1.9825 (0.8061-4.8760)	2.3572 (1.0094-5.5042)*
Glucan		
Chest tightness	3.5350 (0.7480-16.7059)	5.2799 (1.2653-22.0322)*
Cough with phlegm	2.2844 (0.8578-6.0836)	2.6736 (1.0829-6.6007)*
Hoarse/Parched Throat	2.4026 (0.8699-6.6357)	3.5217 (1.3430-9.2350)*
Stomach problems	2.3113 (0.4752-11.2422)	5.7389 (1.4465-22.7692)*
Irritated nose/sneeze	2.4125 (1.0246-5.6805)*	0.8899 (0.4133-1.9162)
Stuffy nose	1.2867 (0.5630-2.9408)	2.3138 (1.0333-5.1810)*

*Indicates significant association between exposure and health (p=0.05)

16

Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	1	Quantificati Consequen Emissions	on of Health ces of	Quantifie Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revi Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	1	Landfill	Tables

Table 3.5 Epidemiological studies of potential health effects associated with composting sites

		Asses	ssment		Association of diagnosed or		
Author	Study subjects	Exposure	Health	Self-reported health complaints	reported health effect or self- reported health complaint with exposure		
(Wheeler et al., 2001)	 13 workers at 3 composting sites 	Aerosol concentrations of total bacteria, fungi, Gram- negative bacteria, streptococcus and actinomycetes; inhalable dust, VOCs, odours and noise.	Symptom questionnaire, physical examination, respiratory function tests, spirometry tests, blood samples and urinalysis	 Flu-type symptoms Gastro-intestinal symptoms Itchy arms and eyes Dry throat and cough 	 No significant association reported by researchers 		
(Bunger <i>et al.</i> 2000)	 58 Compost workers Control group of 40 newly employed workers 	Serum concentrations of specific immunoglobulin G antibodies to moulds and bacteria as immunological markers for bioaerosols.	Physical examination and symptom questionnaire	◆ Nausea	 Significant association of exposure to bioaerosols with diagnosed tracheobronchitis. Significant association of exposure to bioaeosols with frequency of health complaints Also diagnosed but not significant: mucous membrane irritation, sinusitis, eczema, dermat-omycosis, pyoderma, and otitis externa 		
(Douwes <i>et al.</i> 1997)	 14 Compost workers Control group of 10 university students and staff 	Dust, endotoxin, total bacteria and Gram- negative bacteria	Pre- and post-shift nasal lavages for cell types and humoral biomarkers of inflammation		• Some association of Acute and (sub-) chronic non-immune or type III allergic inflammation in upper airways with endotoxin exposure		



Introduction	Review of Informati on Emissions	on Review of Epidemiologica Research	I	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

Table 3.5: Epidemiological studies of potential health effects associated with composting sites continued

		Asses	ssment		Association of diagnosed or		
Author	Study subjects	Exposure	Health	complaints	reported health complaint with exposure		
(Douwes et al. 2000)	 14 Compost workers from previous study 15 Compost workers Control group of 10 university students and staff 	Dust, endotoxin, glucans, total fungi, total bacteria and Gram-negative bacteria	Pre- and post-shift nasal lavages for cell types and humoral biomarkers of inflammation Questionnaire on respiratory symptoms	 Frequent sneezing, and runny/stuffy nose Wheezing and chest tightness 	 Some association of Acute and (sub) chronic non-immune or type III allergic inflammation in upper aiways with endotoxin and (1 → 3)-β-D-glucan exposure No clear association between biomarkers and self-reported respiratory symptoms 		
(Marth <i>et al.,</i> 1997)	 Compost workers Control group of 20 office workers 	Length of employment used as a marker for exposure	Total immunoglobulin E and mold-allergen specific immunoglobulin E concentrations Spirometry Symptom questionnaire	 Asthma Allergy Nausea Respiratory problems Diarrhoea Eczema Eye-irritation Flu-like symptoms Articulation problems Skin problems Cough Hoarseness Allergic rhinitis Smell problems 	 No significant impairment of lung function of employees determined No significant difference between immunoglobulin E concentrations of employees and control group 		



Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	d	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal ('aste Options	Incineration	ı	Landfill	Tables

Table 3.5: Epidemiological studies of potential health effects associated with composting sites continued

		Asse	ssment		Association of diagnosed or		
Author	Study subjects	Exposure	Health	complaints	reported health complaint with exposure		
(Ivens <i>et al,</i> . 1997a; 1997b; 1999)	 28 compost plant workers in initial study 1747 waste collectors control group of 1111 municipal outdoor workers 	Bioaerosol exposure described by viable fungi, total count of fungal spores, microorganisms, and endotoxins	Symptom questionnaire	 Nausea Diarrhoea 	 Working at compost plant was associated (but not significantly) with diarrhoea Significant association between endotoxin and viable fungi exposure with diarrhoea and between endotoxin exposure with nausea for waste collectors with very high exposure 		
(Cobb <i>et al.,</i> 1995)	 Public within 900 m of mushroom compost processing plant 				 No significant association of health hazard with compost plant 		
(Browne <i>et</i> <i>al.,</i> 2001)	 63 persons living close to a grass and leaf composting plant control group of 82 	Bioaerosol concentrations measured in vicinity of plant	Symptom diary		 No significant increase in allergy and asthma symptoms in people living close to the plant, although residents were exposed to elevated concentrations of A. fumigatus as a result of the operations of the compost facility 		
(Sigsgaard et <i>al.,</i> 1997)	 8 compost plant workers control group of 119 drinking water supply workers 	Total dust including endotoxin, airborne bacteria and fungi, sera and blood concentrations of trace elements	Symptom questionnaire Skin prick test for 10 common inhalant allergens, storage mites and moulds		 Gastrointestinal symptoms, ever having experienced vomiting or diarrhoea in relation to work, was significantly more common in the compost workers than the control group 		



Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	al	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revie Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	n	Landfill	Tables

Table 3.5: Epidemiological studies of potential health effects associated with composting sites continued

		Asse	ssment		Association of diagnosed or
Author	Study subjects	Exposure	Health	complaints	reported health effect or self- reported health complaint with exposure
(Herr <i>et al.</i> , 2003)	356 persons living in vicinity of a composting site	Concentrations of total bacteria, moulds and thermophilic and thermotolerant actinomycetes measured near site	Symptom questionnaire	 Colds Hay fever Sinusitis Bronchitis Pneumonia Other respiratory complaints Itching eyes Smarting eyes Loss of appetite Nausea Diarrhoea Excessive tiredness Shivering Fever Joint trouble Muscular complaints 	 Positive associations were reported for residency within 150 – 200 m from the site compared with the unexposed controls for the self reported health complaints: Waking up due to coughing Coughing on rising or during day Bronchitis Excessive tiredness Reports of irritative airway complaints were associated with residency in the highest bioaerosol exposure, 150 – 200 m from the site, and period of residency more than 5 years


Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	Review of Qu Epidemiological Co Research En		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Com	posting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

Table 3.6 VOC Emissions from MBT composting (from Lahl et al., 1998)

Compound	MBT characteristics μg m ⁻³
benzene	0.068
toluene	0.82
Ethylbenzene	2.8
mp-xylene	4.5
styrene	0.46
o-xylene	2.3
acetone	12.8
2-butanone	5

Introduction	Review of Informati on Emissions	formation s Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	erials Recycling Facilities Com		nposting Other W Disposal C		Incineration	ı	Landfill	Tables

Table 3.9Summary of studies relating to health effects of incineration (from Environment Agency, 2003l; draft)
(a) epidemiological studies of cancer risk of waste incineration (WI) on community residents

Study	Study subjects/Type of Incinerator	Exposure Assessment	Outcomes	Results
Diggle <i>et</i> <i>al.</i> , 1990 Gatrell and Lovett, 1992	58 laryngeal cancer cases registered in Lancashire, Great Britain, 1974–1983 Control: 978 lung cancer cases Industrial WI used for disposal of solvents and oils, 1972– 1980	Distance from residence to WI	Incidence of laryngeal cancer	Statistically significant associations with proximity to WI (excess RR = 25.26 at the WI for laryngeal cancer cases compared to lung cancer controls; decay in excess RR = 0.952 per km removed from WI; likelihood ratio test for no excess risk with proximity to WI p < 0.01) Significance remained when alternative controls used
Elliott <i>et al.</i> , 1992	Residents near 10 incinerators of waste solvents and oils in Great Britain	All study areas Distance from the incinerator (<3, 3-10 km)	Incidence of cancers of larynx and lung	No excess in incidence of cancers of larynx and lung compared to national rates Cancer of the larynx: within 3 km of WI O/E = 1.04 with five-year lag (95% CI 0.77, 1.38) and O/E = 1.08 with ten-year lag (95% CI 0.66, 1.67) Cancer of the lung: within 3 km of WI O/E = 0.97 with five-year lag (95% CI 0.91, 1.03) and O/E = 0.94 with ten-year lag (95% CI 0.84, 1.04) No significant increase in cancer risk with closer proximity to the incinerators Cancer of the larynx: overall likelihood ratio 0.48 with five-year lag ($p = 0.77$) and 1.90 with ten-year lag ($p = 0.34$) Cancer of the lung: overall likelihood ratio 0.63 with five-year lag ($p = 0.74$) and 2.3 with ten- year lag ($p = 0.24$)
Biggeri <i>et</i> <i>al.</i> , 1996	755 male lung cancer cases and 755 matched (date of death, sex, age) controls in Trieste, Italy 4 sources: shipyard, iron foundry, city centre, a WI	Spatial models based on distance from each of the four sources	Lung cancer deaths	Lung cancer risk significantly related to the incinerator: excessive RR = 6.7 in the source, adjusting for age, smoking, air particulates, and occupational carcinogens Decay in excess RR = 0.176 per m removed from WI; likelihood ratio test for no excess RR with proximity to WI p = 0.0098)



Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	I	Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	aste Options	Incineration	ı	Landfill	Tables

Table 3.9Summary of studies relating to health effects of incineration (from Environment Agency, 20031; draft)
(a) epidemiological studies of cancer risk of waste incineration on community residents continued

Study	Study subjects/Type of Incinerator	Exposure Assessment	Outcomes	Results
Elliott <i>et al.</i> , 1996, 2000 *	People living within 7.5 km of 72 municipal solid Wis in Great Britain Stage 1: stratified random sample of 20 Wis (stratified by size of local population) Stage 2: remaining 52 Wis	Distance from the incinerator (0.5, 1, 2, 3, 4.6, 5.7, 6.7, 7.5 km)	Incidence of all and selected cancers	Both stages of the study: risk of all cancers, and of stomach, colorectal, liver, and lung cancer decreased as distance increased; residual confounding All cancers: within 3 km of WI with ten-year lag O/E = 1.08 in Stage 1 (95% CI 1.07, 1.10) and O/E = 1.04 in Stage 2 (95% CI 1.03, 1.04) Stomach cancer: within 3 km of WI with ten-year lag O/E = 1.07 in Stage 1 (95% CI 1.02, 1.13) and O/E = 1.05 in Stage 2 (95% CI 1.03, 1.08) Colorectal cancer: within 3 km of WI with ten-year lag O/E = 1.11 in Stage 1 (95% CI 1.07, 1.15) and O/E = 1.04 in Stage 2 (95% CI 1.02, 1.06) Liver cancer: within 3 km of WI with ten-year lag O/E = 1.29 in Stage 1 (95% CI 1.01, 1.51) and O/E = 1.13 in Stage 2 (95% CI 1.05, 1.22) Lung cancer: within 3 km of WI with ten-year lag O/E = 1.14 in Stage 1 (95% CI 1.11, 1.17) and O/E = 1.07 in Stage 2 (95% CI 1.07, 1.09) Revised estimates of excess incidence of primary liver cancer obtained after validation of original diagnoses
Michelozzi <i>et al.</i> , 1998 *	Residents within 10 km of a waste disposal site, a municipal WI, and an oil refinery plant in Rome	Distance from the sources (0-3, 3-8, 8-10 km	Deaths by cancer of liver, larynx, lung, kidney and lymphatic and haematopoietic systems in 1987– 1993	No associations between cancer deaths and distance, except for laryngeal cancer in men – with a significant decline as distance increased All cancers: within 3 km of sources and with adjustment for deprivation SMR = 88 in men (95% Cl 60, 126) and SMR = 96 in women (95% Cl 57, 150) Liver cancer: within 3 km of sources and with adjustment for deprivation SMR = 0 in men and women (no cases observed) Laryngeal cancer: within 3 km of sources and with adjustment for deprivation SMR = 236 in men (95% Cl 27, 800) and SMR = 0 in women (no cases observed) Lung cancer: within 3 km of sources and with adjustment for deprivation SMR = 95 in men (95% Cl 48, 169) and SMR = 55 in women (95% Cl 2, 271) Kidney cancer: within 3 km of sources and with adjustment for deprivation SMR = 276 in men (95% Cl 31, 934) and SMR = 0 in women (no cases observed) Lymphatic and haematopoietic cancers: within 3 km of sources and with adjustment for deprivation SMR = 120 in men (95% Cl 24, 337) and SMR = 64 in women (95% Cl 3, 317) Significance of association between proximity to sources and laryngeal cancer removed by adjusting for deprivation (likelihood ratio tests for declining male mortality with distance from sources p = 0.03 without adjustment for deprivation and p = 0.06 with adjustment for deprivation)



Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	I	Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	ı	Landfill	Tables

Table 3.9Summary of studies relating to health effects of incineration (from Environment Agency, 2003l; draft)
(a) epidemiological studies of cancer risk of waste incineration on community residents continued

Study	Study subjects/Type of Incinerator	Exposure Assessment	Outcomes	Results
Knox, 2000 *	9224 childhood cancer mortality cases in Great Britain who had moved house between birth and death 70 municipal and 307 hospital Wis in Great Britain	Migration towards or away from Wis	Childhood cancers/leukaemias	Highly significant excesses of migrations away from birthplaces close to municipal WIs (RR = 1.27 for distances ≤ 3 km; p < 0.05) Hospital incinerators gave analogous results (no effect sizes reported)
Viel e <i>t al.</i> , 2000 *	Soft-tissue sarcoma and non- Hodgkin's lymphoma cases in 26 electoral wards (1980– 1995) in Doubs, France near a municipal solid WI which began operating in 1971 Control: Hodgkin's disease	Proximity of cancer clusters to the WI	Incidence of soft- tissue sarcoma and non-Hodgkin's lymphoma	Strongest evidence of clustering found in 2 electoral wards near the WI Soft-tissue sarcomas: SIR = 1.44 (p = 0.004) Non-Hodgkin's lymphomas: SIR = 1.27 (p < 0.001) No significant clustering of Hodgkin's disease
Floret et al., (2003) *	222 incident cases of non- Hodgkin lymphoma diagnosed between 1980 and 1995 and controls selected from the 1990 population census using a 10:1 match	Three exposure categories derived from modelling dispersion of dioxin emissions	Non-Hodgkin's Lymphoma	For exposure groups: Low; OR = 1.3 (CI = 0.8-2.0) Medium; OR = 1.0 (CI = 0.6-1.6) High; OR = 2.1 (CI = 1.1-3.7)

CI, confidence interval; O/E, ratio of observed to expected cases; RR, relative risk; SIR, standardised incidence ratio; SMR, standardised mortality ratio; WI, waste incinerator

Adapted from Hu and Shy (2001) and Environment Agency, 2003l; draft

*: epidemiological study; #: review paper



Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	ı	Landfill	Tables

Table 3.9Summary of studies relating to health effects of incineration (from Environment Agency, 2003l; draft)
(b) epidemiological studies of respiratory effects of waste incineration on community residents

Study	Study subjects/Type of Incinerator	Exposure Assessment	Outcomes	Results
Gray et al., 1994 *	713 children in 2 regions (Manly and Malabar) near 2 sludge burning WIs in Sydney Controls: 626 children in a region with no WI	Assessment Air monitoring Region of residence	Prevalence of respiratory illness, airway hyperresponsiveness, atopy; FEV ₁	No significant differences in baseline FEV ₁ and prevalence of current asthma, atopy, symptom frequency, or severity of asthma illness between study and control regions Baseline FEV ₁ : mean values of 1.81 in Manly (95% CI 0.5, 3.1), 1.78 in Malabar (95% CI 0.4, 3.2) and 1.83 in control region (95% CI 0.8, 2.8) Recent wheeze: prevalence of 20.2% in Manly, 22.3% in Malabar and 22.7% in control region (RR = 0.89 and 0.98) Recent medication use: prevalence of 7.8% in Manly, 9.0% in Malabar and 11.4% in control region (RR = 0.68 and 0.79) Atopy: prevalence of 38.3% in Manly, 41.0% in Malabar and 45.7% in control region (RR = 0.84 and 0.90) Doctor-diagnosed asthma: prevalence of 22.3% in Manly, 21.4% in Malabar and 25.2% in control region (RR = 0.88 and 0.85) Airway hyperresponsiveness: prevalence of 17.3% in Manly, 18.0% in Malabar and 19.2% in
				control region (RR = 0.90 and 0.94)
				control region (RR = 0.90 and 0.94) Current asthma: prevalence of 10.2% in Manly, 11.8% in Malabar and 13.1% in control
				region ($RR = 0.78$ and 0.90)



Introduction	Review of Informati on Emissions	on Review of Epidemiologica Research	w of Quantificati emiological Consequen arch Emissions		on of Health ces of Quantific Environr Consequ Emission		ification of the H Inmental H iquences of R ions F		ext for Quantified th and conmental Risks, & ew of Public eption Issues	Conclusions	
Glossary	Introduction	Materials Recycling Facilities	erials Recycling Facilities Corr		nposting Other W Disposal C		Incineration	ı	Landfill	Tables	

Table 3.9Summary of studies relating to health effects of incineration (from Environment Agency, 2003l; draft)
(b) epidemiological studies of respiratory effects of waste incineration on community residents continued

Study	Study subjects/Type of Incinerator	Exposure Assessment	Outcomes	Results
Shy <i>et al.</i> , 1995 * Mohan <i>et al.</i> , 2000 * Hu <i>et al.</i> , 2001 *	Selected normal and sensitive residents of 3 Wis (biomedical, hazardous, and municipal) and 3 matched comparison communities in NC, and a hazardous WI and matched comparison community in SC, 1992-1994	Air monitoring; wind sector analysis; chemical mass balance receptor modelling; dispersion modelling	Prevalence of acute/chronic respiratory diseases/symptoms; PEFR, FEV ₁ ; cell counts, albumin, etc, in nasal lavage FVC and FEF _{25-75%}	No consistent differences in prevalence of respiratory symptoms between matched communities, adjusting for age, sex, race, education, respiratory disease risk factors (effect sizes not reported) No consistent differences in nasal lavage analysis (effect sizes not reported) Normal subjects: percentage of predicted FEV, and PEFR higher in all WI communities than comparison communities (not statistically significant) Among sensitives: statistically significant difference in percentage of predicted PEFR between two matched pairs of communities (100.7% and 116.2% in biomedical and municipal WI communies respectively, versus 112.4% and 100.8% in comparison communities; $p < 0.05$ and $p < 0.05$) Wheeze: adjusted OR = 1.2, 0.9 and 1.1 for biomedical, municipal and hazardous Wis in North Carolina ($p > 0.05$, $p > 0.05$, $p > 0.05$), and adjusted OR = 1.5 for hazardous WI in SC ($p < 0.05$) Morning cough/wheeze with phlegm: adjusted OR = 1.1, 1.0 and 1.1 for biomedical, municipal and hazardous Wis in NC ($p > 0.05$, $p > 0.05$, $p > 0.05$, $p > 0.05$, and adjusted OR = 1.7 for hazardous WI in SC ($p < 0.05$) Awakened at night with respiratory symptoms: adjusted OR = 0.8, 1.0 and 1.3 for biomedical, municipal and hazardous WI in NC ($p < 0.05$) Two or more short duration symptoms: adjusted OR = 0.7, 1.2 and 1.1 for biomedical, municipal and hazardous WI in NC ($p < 0.05$, $p > 0.05$, $p > 0.05$), and adjusted OR = 1.6 for hazardous WI in 1993 (difference of -8.62% ; $p < 0.05$) and the municipal WI in 1994 (difference of -1.29% ; $p < 0.05$) Sensitive subjects no more adversely affected than hay fever sufferers or normal subjects (effect sizes not reported)



Introduction	Review of Informati on Emissions	ion Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions	
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	ı	Landfill	Tables	

 Table 3.9
 Summary of studies relating to health effects of incineration (from Environment Agency, 2003l; draft)

 (b) epidemiological studies of respiratory effects of waste incineration on community residents continued

Study	Study subjects/Type of Incinerator	Exposure Assessment	Outcomes	Results
Legator et al., 1998 * Pichette, 2000	58 residents living within 5 km of cement kiln toxic Wis in Midlothian, Texas Control: 54 residents in a nearby community with similar demographic characteristics	Region of residence	Prevalence of symptoms covering 12 categories of physical health/function, including respiratory symptoms	Respiratory symptoms significantly more prevalent in Midlothian residents (OR = 5.15, 95% CI 1.67, 19.63) Ears, nose and throat: OR = 1.23 (95% CI 0.55, 2.78) Central nervous system: OR = 1.39 (95% CI 0.61, 3.22) Cardiovascular system: OR = 0.66 (95% CI 0.29, 1.49) Muscles and bone: OR = 0.49 (95% CI 0.20, 1.12) Immune system: OR = 1.35 (95% CI 0.52, 3.62) Skin: OR = 1.51 (95% CI 0.57, 4.17) Urinary system: OR = 1.04 (95% CI 0.35, 3.11) Endocrine system: OR = 0.82 (95% CI 0.28, 2.32) Digestive system: OR = 1.18 (95% CI 0.39, 3.70) Teeth and gums: OR = 4.42 (95% CI 0.69, 139.56) Blood: OR = 0.93 (95% CI 0.14, 6.10) Limitations of study include small sample size, potential for bias, and limited control for confounding variables

CI, confidence interval; FEF25-75%, forced expiratory flow; FEV1, forced expiratory volume in 1 sec; FVC, forced vital capacity; OR, odds ratio; PEFR, peak expiratory flow rate; RR, relative risk; WI, waste incinerator

Adapted from Hu and Shy (2001) and Environment Agency, 2003l; draft

*: epidemiological study; #: review paper



Introduction	Review of Informati on Emissions	ion Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions	
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	ı	Landfill	Tables	

Table 3.9Summary of studies relating to health effects of incineration (from Environment Agency, 2003l; draft)
(c) epidemiological studies of the reproductive effects of waste incineration on community residents

Study	Study subjects/Type of incinerator	Exposure Assessment	Outcomes	Results
Lloyd et al., 1988 *	Residents in areas exposed to a municipal and a chemical WI Control: residents in neighbouring areas	By postcode sectors and wind direction	Frequency of twinning in Scotland, 1976–1979 and 1980–1983	Frequency of twinning increased in areas at most risk from the emissions $1976-1979$: 7.3 and 12.4 twins per 1000 births in high risk areas compared with 7.1 per 1000 in background areas (RR = 1.0 and 1.7)
1989				1980–1983: 16.0 and 19.9 twins per 1000 births in high risk areas compared with 7.6 per 1000 in background areas (RR = 2.1 and 2.6) Limitations of the study and interpretation of results discussed in published
				correspondence
Jansson and Voog,	Case study (CS): 6 children with cleft lip and palate	ft lip CS: Incidence of cleft lip and palate malformations in		CS: highest levels of dioxin within 1 km of the WI, decreased as distance increased
1989 *	Register study (RS): 18 boroughs with municipal Wis	dispersal calculation of dioxin exposure	Sweden, 1973–1986	CS and RS: no increased risk of cleft lip and palate in studied areas after start of incineration
		RS: before and		RS before incineration: 1.81 cases of cleft lip and palate per 1000 births
		after start of Wis		RS after incineration: 1.85 cases per 1000 births (RR = 1.02; 95% CI 0.71, 1.47)
Williams et al., 1992 *	Residents in at-risk areas near two Wis	By wind speed/direction,	Male/female sex ratio of births in Scotland, 1975–	No differences between the at-risk and comparison areas (male/female ratios of 1.01 and 0.99)
	Residents in a comparison area	local topography, soil levels of pollutants	1979 and 1980–1983	A significant excess of female births in the district at most risk (male/female ratio of 0.87; p < 0.05)
Rydh-	Residents in all municipalities or in 14	With or without	Incidence of twinning,	No clustering of twinning in area/time
stroem, 1998 *	municipalities near 14 refuse Wis in Sweden	WI; before and after start of WI	1973–1990	RR (before vs. after) significantly increased in one municipality with WI (RR = 1.72; 95% CI 1.22, 2.43) but significantly decreased in one of the other municipalities with WI (RR = 0.46; 95% CI 0.29, 0.73)
Ten Tusscher	8803 births at Zeeburg maternity clinic, Amsterdam, The Netherlands near a	Proximity of clinic to WI site	Incidence of orofacial clefts	Incidence of orofacial clefts at Zeeburg clinic, 1960–1969 was 2.5 per 1000 births
<i>et al</i> , 2000 [*]	chemical waste, 1961–1973			Incidence at comparison clinic was 1.2 per 1000 births (RR = 2.1)
	Control: 21078 births at a maternity clinic further away from the WI site			RR greater during 1963–1964 (effect size not reported)



Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	l	Quantification Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	aste Options	Incineration	1	Landfill	Tables

Table 3.9Summary of Studies Relating to Health Effects of Incineration (from Environment Agency, 2003l; draft)(c) epidemiological studies of the reproductive effects of waste incineration on community residents continued

Study	Study subjects/Type of incinerator	Exposure Assessment	Outcomes	Results
Dummer et al., 2003 *	244 758 births to mothers living in Cumbria where 3 incinerators deal with material classified as 'difficult' by the Environment Agency, 1 incinerator processes only inert and biodegradable material, and 3 crematoriums	Proximity of mothers' address to incinerators and crematoriums	Incidence of stillbirth, neonatal death and lethal congenital anomalies amongst babies	Increased risk of lethal congenital anomaly, in particular spina bifida (RR = 1.17 ; 95% CI:1.07 to 1.28) and heart defects (RR = 1.12 ; 95% CI: 1.03 to 1.22) around incinerators and increased risks of stillbirth (RR = 1.04 ; 95% CI: 1.01 to 1.07) and anencephalus (RR = 1.05 ; 95% CI: 1.00 to 1.10) around crematoriums

CI, confidence interval; RR, relative risk; WI, waste incinerator

Adapted fron Hu and Shy (2001) and Environment Agency, (2003l; draft)

*: epidemiological study; #: review paper



Introduction	Review of Information	on Review of Epidemiologica Research	I	Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		Cont Heal Envir Revi Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

 Table 3.17
 Single-site epidemiological studies of potential health effects associated with landfill sites (from Redfearn and Roberts, 2002)

Site	Type of Waste Received/Years of Operation/Other Site Details	Primary Exposure Route / Chemicals of Concern	Reason for Initiation of Study and/or Site Closure	Area or Population Treated As Being Exposed / Study Period	Health Effects Examined	Presence of Association With Exposure
Nant-y- Gwyddon Iandfill, Wales (<i>Fielder et al.</i> 2000a, 1997, 2000b; Mukerjee & Deacon 1999; Richardson 1999)	household; industrial; commercial; difficult 1988 – present	landfill gas	community concerns that odours from site causing a variety of conditions	Fielder <i>et al.</i> 2000a, 1997 residents in 5 electoral wards within 3km of site 1981 – 1997	Mortality: all causes; respiratory disease; cancers Hospital admissions: general admissions; respiratory disease; asthma; cancer; sarcoidosis; spontaneous abortions Low birth weight Birth defects: all anomalies abdominal wall (gastroschisis) Drug prescription rates for gastrointestinal, respiratory and central nervous systems, skin and eyes	Non-significant Non-significant Non-significant Significant positive Significant positive Elevated
				Fielder et al. 2000b residents in 5 electoral wards within 3km of site 1998-2000	Time to pregnancy	Non-significant
				Mukerjee & Deacon 1999 residents within 1km, 1-2km, 2- 3km and >3km from site 1998	Self reported symptoms: headache, sore throat, runny nose, feeling sick, diarrhoea Self reported symptoms: sore eyes, dizziness, skin rash Self-reported chronic diseases Frequency of GP consultations	Elevated Non-significant Non-significant Non-significant
				Richardson 1999 residents in 5 electoral wards within 3km of site 1991-1998	Sarcoidosis	Elevated



Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	Review of Epidemiological Research		Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions		ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

Table 3.17: Single-site epidemiological studies of potential health effects associated with landfill sites (from Redfearn and Roberts, 2002) continued

Site	Type of Waste Received/Years of Operation/Other Site Details	Primary Exposure Route / Chemicals of Concern	Reason for Initiation of Study and/or Site Closure	Area or Population Treated As Being Exposed / Study Period	Health Effects Examined	Presence of Association With Exposure
Lipari landfill, New Jersey (Berry & Bove 1997)	municipal; household; liquid and semi-solid chemical; other industrial 1958 - 1971 ranked no. 1 on US EPA's National Priority List liquid wastes emptied from containers prior to disposal hazardous leachate migrated into nearby streams and a lake immediately adjacent to community with homes, schools and playgrounds	Inhalation of volatilised chemicals emitted from landfill and from contaminated waters	public complaints regarding odour, respiratory problems, headaches, nausea and dying vegetation	radius of 1km from perimeter of site, including high exposure group adjacent and downwind of site 1961 – 1985	Average birth weight Proportion low birth weight Proportion premature births	Significant positive Significant positive Significant positive
Miron Quarry, Quebec; (Goldberg et al. 1995a, 1995b, 1999)	domestic; industrial; commercial 1968 - present 3rd largest municipal solid waste landfill site in North America 100,000 people live within 2km has not been capped biogas collection system installed in 1980, and operated at low efficiency	release of landfill gas into ambient air and soil	health concerns expressed by local residents; frequent odour complaints registered	<u>Goldberg et al. 1995a</u> postal code areas containing and bordering site (up to 4km from perimeter of site) 1979 – 1989 <u>Goldberg et al. 1995b</u> postal code areas containing and bordering site (up to 4km from perimeter of site) 1981 - 1988 <u>Goldberg et al. 1999</u>	Low birth weight Very low birth weight Small for gestational age Preterm births Males: cancers of stomach; liver & intrahepatic bile duct; trachea, bronchus & lung; prostate Females: cancer of stomach; cervix uteri Females, breast 13 other cancer sites in males; 17 other cancer sites in females	Significant positive Non-significant Significant positive Non-significant Significant or nearly significant or nearly significant or nearly significant positive Significant negative No association
				postal code areas containing and bordering site (up to 4km from perimeter of site) 1979 – 1985	Males: cancer of liver; kidney; pancreas; prostate; and non-Hodgkin's lymphomas 8 other cancer sites in males	Significant or nearly significant positive No association

Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	d	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	ı	Landfill	Tables

Table 3.17: Single-site epidemiological studies of potential health effects associated with landfill sites (from Redfearn and Roberts, 2002) continued

Site	Type of Waste Received/Years of Operation/Other Site Details	Primary Exposure Route / Chemicals of Concern	Reason for Initiation of Study and/or Site Closure	Area or Population Treated As Being Exposed / Study Period	Health Effects Examined	Presence of Association With Exposure
BKK landfill, California (Kharrazi e<i>t</i> <i>al.</i> 1997)	hazardous waste of all types; municipal 1963 – 1989 received nearly 4 million tons of hazardous waste residential developments in close proximity numerous complaints of odour, surface water runoff onto nearby streets, hazardous waste spills from HGVs, and dust releases	airborne exposures	concerns over public health and welfare following complaints of odours, surface water runoff, hazardous waste spills from trucks and dust releases	residence in areas with high rates of odour complaints (high odour area up to 0.6 miles from landfill) 1978 – 1986	Reduction in gestational age Low mean birth weight Fetal and infant mortality	Significant positive Significant positive No association
Montchanin landfill, France (Zmirou <i>et al.</i> 1994,	liquid and solid toxic industrial, including wastewater treatment sludge, dehydrated hydroxide sludge and solvent-containing wastes 1979 – 1988	VOCs in ambient air	community health concerns triggered by offensive odours, suspected increase in certain health	Zmirou <i>et al.</i> 1994 estimated exposures using air dispersion model 1987 – 1989	Drug consumption rates for respiratory, opthalmological, dermatological, gastro- intestinal and neurological conditions	No significant association
Deloraine et al. 1995)	received 400,000 tons of industrial wastes located adjacent to town of 6000 inhabitants - 100m from nearest houses		complaints, and elevated levels of VOCs in ambient air	Deloraine <i>et al.</i> 1995 estimated exposures using air dispersion model 1990	Psychiatric disorders Respiratory symptoms Isolated biological abnormalities Skin diseases Eye diseases Ear, nose and throat conditions Miscellaneous conditions	Significant positive Significant positive No association No association No association No association No association
Upper Ottawa Street landfill, Ontario (Hertzman et al. 1987)	solid and liquid industrial; commercial; domestic 1950s - 1980 volumes of industrial waste received increased throughout 1970s, such that approx. 8 to 12 million gallons of liquid waste disposed of during 1978 capped in 1980/81	airborne exposures to vapours, fumes, dust or ash, as well as direct skin contact	public concerns regarding health effects	residence within 750m from edge of tipping face approx. 1984	Self-reported respiratory, skin, mood, narcotic and eye conditions Self-reported muscle weakness Self-reported adverse birth outcomes: low birth weight stillbirth miscarriage/spontaneous abortion birth defects	Significant positive No association No association No association No association No association
Waste disposal site, Northwestern Illinois (Mallin 1990)	municipal; industrial, including solvents, plating wastes and heavy metals late 1950s – 1972	Drinking water from wells contaminated with VOCs	several areas of elevated mortality from bladder cancer identified in region	residence in town using water from contaminated wells 1977-1985	Bladder incidence	Significant positive

Introduction	Review of Informati on Emissions	ion Review of Epidemiologica Research	al	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revi Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	1	Landfill	Tables

Table 3.18: Multiple site epidemiological studies of potential health effects associated with waste disposal sites (from Redfearn and Roberts, 2002)

Author (s)	Study Parameter s	Type of Sites Evaluated/ Years of Operation	Area or Population Treated As Being Exposed/ Study Period	Health Effects Examined	Presence of Association With Exposure
Dolk <i>et al.</i> 1998	21 landfill sites in 5 European Countries	landfill sites handling hazardous chemical wastes majority either opened before mid-1970s or closed before mid- to late- 1980s	maternal residence within 3 km of landfill site mid/late 1980s – 1993 in most cases	Non-chromosomal birth defects: all anomalies; neural tube; cardiac septa; great arteries and veins Non-chromosomal birth defects: tracheo- oesophageal; hypospadias; gastroschisis 19 other specified types of non-chromosomal birth defects	Significant positive Nearly significant positive No association
Vrijheid <i>et al.</i> 2002	23 Iandfill sites in 5 European Countries	landfill sites handling hazardous chemical wastes majority either opened before mid-1970s or closed before mid- to late- 1980s	maternal residence within 3 km of landfill site mid/late 1980s – 1993 in most cases	Chromosomal birth defects	Significant positive
Elliott <i>et</i> <i>al.</i> 2001a, 2001b	9,565 landfill sites in England, Wales and Scotland	774 special waste landfills, 7,803 non- special waste landfills, and 988 classified as unknown sites operational between 1982 and 1997	residence within 2 km of landfill site 1983 – 1998	Birth defects: all anomalies; neural tube; hypospadias/epispadias; abdominal wall; gastroschisis/exomphalos Birth defects: cardiovascular Low and very low birth weight Still births Cancer registrations: bladder; brain; hepatobiliary; childhood and adult leukaemia	Excess risks Depressed risks Excess risks No association No association
Lewis- Michl et <i>al.</i> 1998	38 landfill sites in New York State, USA	municipal landfills with soil-gas migration conditions; selected from the New York State Inactive Hazardous Waste Site Registry; sites in NY City excluded majority of landfills opened prior to 1970, closed prior to end of 1980s majority not capped or lined	residence within 250 ft of landfill site boundary (or greater distance if further gas migration shown) 1980 – 1989	Male cancer incidence: liver; lung; bladder; kidney; brain; non-Hodgkin's lymphoma; leukaemia Female cancer incidence: liver; lung; kidney; brain; non-Hodgkin's lymphoma Female cancer incidence: bladder; leukaemia	No association No association Significant positive



Introduction	Review of Informatio on Emissions	n Review of Epidemiologica Research	al	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revie Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal (/aste Options	Incineration	۱	Landfill	Tables

 Table 3.19
 Summary of findings of epidemiological studies at landfill sites (from Redfearn and Roberts, 2002)

Health Outcome	Number of Studies Indicating Excess Risks	Number of Studies Indicating No Excess Risk
Birth Defects:		
All chromosomal anomalies	1	0
All non-chromosomal anomalies	3.6 (before site opened in 2 of these)	1
Central nervous system	0	1
Neural tube defects	2	0
Cleft lip/palate	0	1
Heart and circulatory	1	1
Hypospadias/epispadias	2 (borderline in 1; before site opened in 1)	0
Limb reductions	0	1
Abdominal/Gastroschisis	3.7 (borderline in 1; before site opened in	0
Skin and other integument	1)	1
Tracheo-oesophageal	0	0
Renal	1 (borderline)	1
Urinary tract	0	1
	0	
Other Pregnancy Outcomes:		
Low birth weight/prematurity	4	2
Still births	0	3
Infant mortality	0	1
Spontaneous abortions	0	2
Time to pregnancy	0	1
Cancer:		
All types	0	1
Oesophagus	0	1
Stomach	1	1
	2	2
Trachea/bronchus/lung	1	2
Prostate	2	0
	1	0
Breast	0	1
Colorectum	0	2
Brain	0	3
Pancreas	1	1
Kidney	1	2
Bladder	2	3
Leukaemia	1	2
Non-Hodgkin's lymphoma	1	2
Skin melanoma	0	1
Respiratory		
All respiratory diseases	1	
Asthma	0	1
Sarcoidosis	 1 (before industrial tipping commenced) 	1

186

Introduction	Review of Informat on Emissions	ion Review of Epidemiologic Research	al	Quantificati Consequer Emissions	on of Health aces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Cont Heal Envir Revie Perc	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Cor	nposting	Other W Disposal (/aste Options	Incineration	۱	Landfill	Tables

Table 3.19 Summary of findings of epidemiological studies at landfill sites (from Redfearn and Roberts, 2002)

Health Outcome	Number of Studies Indicating Excess Risks	Number of Studies Indicating No Excess Risk
Psychiatric disorders	1	0
Miscellaneous self-reported symptoms	2	0
Drug prescription rates, miscellaneous symptoms	1	1
Unspecified:		
Hospital admissions, all diseases	0	1
Mortality, all causes	0	1



Introduction	Review of Informati on Emissions	ion Review of Epidemiologica Research	I	Quantificati Consequen Emissions	on of Health ces of	Quantific Environr Consequ Emission	cation of the nental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified th and ronmental Risks, & ew of Public eption Issues	Conclusions	
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	'aste Options	Incineration	1	Landfill	Tables	

Table 3.20 Risks of congenital anomalies, stillbirths, and low and very low birth weight in populations living within 2 km of a landfill site (all waste types) during operation or after closure compared with those in the reference area (> 2 km from any site) (from Elliott *et al.*, 2001a)

	Near landfi	ll (<2km)	Refere	ence area		Relative risk (99% Cl)
Birth outcome	No. of cases	Rate (per 100,000 births)	No. of cases	Rate (per 100,000 births)	Unadjusted	Adjusted (but not for deprivation)	Adjusted (and for deprivation)
Congenital anomalies (registe	er and terminations	data*)					
All congenital anomalies	90 272	1 550	34 325	1 694	0.92 (0.907 to 0.923)	1.01 (1.00 to 1.02)	1.01 (1.005 to 1.023)
Neural tube defects	3 508	60	1 140	56	1.07 (1.02 to 1.12)	1.08 (1.03 to 1.12)	1.05 (1.01 to 1.10)
Cardiovascular defects	6 723	115	2 716	134	0.86 (0.83 to 0.89)	0.95 (0.92 to 0.98)	0.96 (0.93 to 0.99)
Hypospadias and epispadias†	7 363	247	2 485	240	1.03 (1.00 to 1.06)	1.07 (1.04 to 1.10)	1.07 (1.04 to 1.10)
Abdominal wall defects	1 488	26	448	22	1.16 (1.08 to 1.23)	1.14 (1.06 to 1.22)	1.08 (1.01 to 1.15)
Congenital anomalies (hospita	al admissions)	·					
Hypospadias and epispadias‡	1 503	257	536	268	0.96 (0.90 to 1.02)	-	0.96 (0.90 to 1.02)
Abdominal wall defects	755	40	227	35	1.13 (1.03 to 1.24)	-	1.07 (0.98 to 1.18)
Gastroschisis and exomphalos‡	467	25	126	19	1.26 (1.12 to 1.42)	-	1.19 (1.05 to 1.34)
Stillbirths and birth weight							
Stillbirths	32 271	532	11 200	514	1.04 (1.02 to 1.05)	1.05 (1.03 to 1.06)	1.00 (0.99 to 1.02)
Low birth weight	422 149	7 000	137 958	6 367	1.10 (1.095 to 1.104)	1.11 (1.102 to 1.111)	1.05 (1.047 to 1.055)
Very low birth weight	62 191	1 031	20 858	963	1.07 (1.06 to 1.08)	1.08 (1.07 to 1.09)	1.04 (1.03 to 1.05)

* Terminations included for England and Wales 1992-8, Scotland 1988-94

† Excludes terminations (3 cases)

[‡] Surgical corrections. Surgical correction of gastroschisis and exomphalos was included in the study of Elliot et al. as a cross-check on the data for abdominal wall defects. The cases in this category are also included in the wider category of abdominal wall defects.

Introduction	Review of Informat on Emissions	ion Review of Epidemiologica Research	I	Quantificati Consequen Emissions	on of Health ces of	Quantifi Environ Conseq Emissio	cation of the mental uences of ns	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Glossary	Introduction	Materials Recycling Facilities	Con	nposting	Other W Disposal C	/aste Options	Incineration	n	Landfill	Tables

Table 3.21 Estimated relative risks (99% confidence Intervals) of birth outcomes for populations living within 2 km of a landfill site, adjusted for deprivation and other variables* according to waste type and to operating status for those sites that opened during the study period (from Elliott et al., 2001a)

	All opera	ting and closed sites, by v	Sites that opened during st operat	udy period (all waste types), by ing status†	
Birth outcome	All wastes	Special waste	Non-special waste	Before opening	During operation or after closure
Congenital anomalies (re	gister and terminations da	ata‡)			
All congenital anomalies	1.01 (1.005 to 1.023)	1.07 (1.04 to 1.09)	1.02 (1.01 to 1.03)	1.02 (0.99 to 1.05)	1.00 (0.99 to 1.01)
Neural tube defects	1.05 (1.01 to 1.10)	1.07 (0.95 to 1.20)	1.06 (1.01 to 1.12)	0.98 (0.82 to 1.16)	1.05 (0.99 to 1.10)
Cardiovascular defects	0.96 (0.93 to 0.99)	1.11 (1.03 to 1.21)	0.95 (0.91 to 0.98)	0.92 (0.80 to 1.04)	0.92 (0.88 to 0.95)
Hypospadias and episadias§	1.07 (1.04 to 1.10)	1.11 (1.03 to 1.21)	1.07 (1.04 to 1.11)	1.08 (0.98 to 1.19)	1.05 (1.02 to 1.09)
Abdominal wall defects	1.08 (1.01 to 1.15)	1.03 (0.86 to 1.25)	1.07 (0.99 to 1.16)	1.24 (0.97 to 1.60)	1.06 (0.98 to 1.14)
Congenital anomalies (ho	ospital admissions)				
Hypospadias and epispadias¶	0.96 (0.90 to 1.02)	0.98 (0.81 to 1.19)	0.96 (0.90 to 1.04)	1.42 (0.94 to 2.16)	0.93 (0.86 to 1.00)
Abdominal wall defects	1.07 (0.98 to 1.18)	1.08 (0.82 to 1.42)	1.05 (0.94 to 1.16)	2.26 (1.23 to 4.15)	1.12 (1.01 to 1.25)
Gastroschisis and exomphalos¶	1.19 (1.05 to 1.34)	1.10 (0.77 to 1.58)	1.18 (1.03 to 1.34)	1.33 (0.46 to 3.81)	1.24 (1.09 to 1.42)
Stillbirths and birth weig	ht				
Stillbirths	1.00 (0.99 to 1.02)	0.99 (0.95 to 1.03)	1.00 (0.99 to 1.02)	1.01 (0.96 to 1.06)	1.02 (1.00 to 1.03)
Low birth weight	1.05 (1.047 to 1.055)	1.05 (1.04 to 1.06)	1.06 (1.052 to 1.062)	1.01 (0.99 to 1.02)	1.07 (1.062 to 1.072)
Very low birth weight	1.04 (1.03 to 1.05)	1.03 (1.00 to 1.06)	1.04 (1.03 to 1.06)	0.98 (0.94 to 1.02)	1.04 (1.03 to 1.05)

† 522 landfill sites with available data for hospital admissions

‡ Terminations included for England and Wales 1992-8, Scotland 1988-94

§ Excludes terminations (3 cases)

¶ Surgical corrections. Note 3: Surgical correction of gastroschisis and exomphalos was included in the study of Elliot et al. as a cross-check on the data for abdominal wall defects. The cases in this category are also included in the wider category of abdominal wall defects.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4. QUANTIFICATION OF THE HEALTH CONSEQUENCES OF EMISSIONS

190

Introduction	Review of Information on Emissions	Review of Quantification of Quantification of Epidemiological Consequences of Cc Emissions Er		Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Summary – quantification of health consequences

We have investigated the health effects of MSW treatment and disposal facilities drawing on information on emissions to air from the previous chapter.

We used an atmospheric dispersion model to evaluate levels of pollution that might be experienced by people living in the vicinity of waste management facilities. We used "dose-response functions" for each substance to estimate the health consequences that would be expected to result from these exposures. We were able to investigate additional deaths, hospital admissions and cancer cases due to exposure to air pollutants. The study was not able to investigate the effects of all emissions to air, or the potential effects of exposure to substances released to water or land.

We estimated the health effects that might arise in the general population due to emissions to air from municipal solid waste management facilities. In view of the margin of uncertainty in these estimates, the presently available data does not allow us to say that one option for managing MSW is definitely better or worse than the other options in terms of the health effects. Although these estimates are of moderate or poor quality, they give an indication of the scale of health effects likely to be associated with emissions to air from waste management activities. We could not investigate the potential health effects of emissions to air from composting facilities in this way because of a lack of information on emissions.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4.1 Scope

This chapter is a quantitative risk assessment, aimed at quantifying some of the potential health effects of waste management facilities on health. The impacts of emissions from waste management facilities on air quality and consequently, on the health of the surrounding population were quantified. As discussed below, this covers only some of the possible health effects associated with waste management facilities. This should be borne in mind when interpreting the results of this study.

The health effects considered are:

- "Deaths brought forward" that is, 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 does not include deaths due to cancer caused by exposure to airborne carcinogens.
- Respiratory hospital admissions
- Cardiovascular hospital admissions
- Additional cases of cancer

Where the effects of a specific emission have not been quantified, this does not necessarily imply an absence of impact. A number of assumptions have been made in carrying out the evaluation, and these draw attention to the key areas of uncertainty. These are detailed within the method section of the chapter and cross-referenced to Chapter 2 and 3 as appropriate.

A staged approach has been adopted for the evaluation of health effects, as outlined in the method section of this chapter. This chapter takes as its starting point work being completed by Birmingham University for the Environment Agency (*Environment Agency, 2003I; draft*). This report has been peer reviewed, and also extensively reviewed by members of the Environment Agency's project board. The Birmingham University study provides a quantification of the health consequences of emissions to air from MSW incineration. The Birmingham University study has been developed to provide similar estimates of health burdens as a result of emissions to air from landfill and other waste management facilities. This comprises modelling of atmospheric dispersion of emissions to air from landfill and other facilities, adopting a parallel approach to that used in the Environment Agency research where applicable.

The study is based on a set of ten conceptual waste management facilities, other than the incineration facility already considered in the Environment Agency study (*Environment Agency, 2003I; draft*)

- Six conceptual landfill facilities, covering small, medium and large size facilities, with flaring of landfill gas, and power generation from the landfill gas;
- a pyrolysis / gasification plant;
- a composting plant;
- an anaerobic digestion plant; and

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

• a mechanical biological treatment (MBT) plant.

A set of assumptions defining the emissions regime have been made (see Section 4.2). Estimated concentration fields have been produced in a comparable way to those produced for MSW incinerator facilities in the research being carried out by Birmingham University for the Environment Agency. Forecasts relating to landfill and the other facilities are subject to greater uncertainty than forecasts for MSW incineration. The predicted concentration field and health burden data is presented in Section 4.3. Comparisons between landfill, incineration and other waste management facilities are made in Section 4.4.

This part of the study is limited in the following ways:

Potential effects due to exposure via pathways other than airborne exposure are not considered. Airborne exposure is likely to be the pathway of greatest concern as regards combustion processes. Other emissions (e.g. to surface water or groundwater) are almost universally treated before consumption – in particular, drinking water is treated before supply and has to comply with strict quality standards. Products spread to land such as compost are now also subject to specific control standards. This reduces the likely significance of these pathways in respect of their potential impact on human health.

Also, the Environment Agency (2003l; draft) study showed that the health effects associated with exposure to dioxins and furans from the thermal treatment processes considered could be discounted. In view of this, indirect exposures via the food chain were not considered in this work.

- Potential effects due to exposure to substances other than those identified in the report are not considered.
- For some processes, no data were available on emissions of potentially significant substances.
- The study assessed the potential effects of emissions under normal operating conditions. For health effects related to long-term exposures (such as those covered in this chapter), this is a valid approach. Short-term increases in emissions under abnormal operating conditions could potentially result in different health effects, although for processes regulated under PPC, any such effects are normally evaluated and controlled as part of the permitting process.

In Section 3.7.3 and Table 3.22, exposure-response functions for adverse birth outcomes are presented, together with comments relating to the quality and reliability of the values. Having considered the advice of Committee on Toxicity and the Royal Society, we conclude that it is not appropriate to apply these exposure-response functions to quantify the significance of the observed associations.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4.2 Method for assessing the health effects of emissions to air

The method developed in the quantification of the health effects of incinerators (*Environment Agency, 2003I; draft*) was used in this study.

Data on emissions were derived from the information given in Chapter 2, supplemented by assumptions on six illustrative landfill sites (this information is set out in Table 4.2 below). This information was used in a dispersion modelling study in order to map typical spatial distributions of ground-level concentrations in the vicinity of the illustrative sites.

Modelled ground level concentrations were assessed in terms of their potential impact on the health of an assumed exposed population.

For some health outcomes, the methodology prepared by the Committee on the Medical Effects of Air Pollutants (COMEAP, 1998) to estimate the incremental effect of emissions of sulphur dioxide, oxides of nitrogen and particulates from the different scenarios on the population was used. The COMEAP method allows the increase in mortality and hospital admissions to be estimated using exposure-response coefficients derived from epidemiological studies. These coefficients represent the percentage increase in a baseline health rate in the population per unit rise in pollutant concentration.

The COMEAP approach was used in preference to other methods, because it represents the most current available method using UK-specific data. The COMEAP approach is used in assessing the health consequences of individual facilities, source groups and policies in the UK.

• The unit risk factors for exposure to chemical genotoxic carcinogens prepared by the World Health Organisation (WHO, 2000) are also used in this study.

The COMEAP and WHO methods and their applicability to this study is described in Chapter 3. It is assumed that the lowest exposures to released substances down to the cut-off level discussed below carry some risk of harm. This is an assumption which cannot be tested in practice, but increases the likelihood that any effects will be over-estimated rather than under-estimated.

The outcome of these calculations is directly related to the size and spatial distribution of the exposed population. Two uniform population densities, typical of urban/suburban and suburban/rural areas of the UK, were adopted and the calculations carried out using these population statistics. These are designed to illustrate typical magnitudes of effect, but will tend to underestimate effects of emission in very highly populated areas, and overestimate effects of emission in areas of sparse population. Landfill sites and composting plants tend to be located in rural areas, whereas the other facilities are likely to be in urban centres.

Both the COMEAP factors and WHO unit risk factors refer to health outcomes which do not exhibit any apparent threshold of effect. It is therefore possible to apply these factors to the very low incremental exposures identified in this study.

In practical terms, the method developed in *Environment Agency (2003l; draft)* can be summarised as follows:

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

- Carry out a dispersion modelling study to identify pollutant levels due to emissions from the process at discrete receptor points across a suitably sized grid;
- Divide the exposed population into individual discrete geographical areas around each receptor point;
- For each area calculate the health effect by multiplying the population by the exposure-response coefficient, the baseline health effect rate (for COMEAP coefficients) and the concentration of the pollutant at that point;
- Sum the health effects over the area studied;
- Repeat for each pollutant and health outcome.

Following the approach adopted in *Environment Agency (2003I; draft)*, the modelling domain was chosen to be that area where the incremental pollution from the source exceeded $0.01\mu g/m^3$ of PM_{10} , SO_2 or NO_2 . The Environment Agency study indicates that an incremental pollutant concentration of $0.01 \ \mu g/m^3$ of PM_{10} , SO_2 or NO_2 is likely to be wholly without effect. It was found that a model domain extending 1 km from the source resulted in incremental concentrations of PM_{10} , SO_2 or NO_2 of less than $0.01 \ \mu g/m^3$. The analysis was carried out at 25m resolution over this area for all modelled emissions.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4.3 **Processes and Substances Considered**

The main concerns regarding landfill site operations generally relate to emissions to the atmosphere from the surface of the landfill and from the combustion of landfill gas.

Emissions from the following processes have been considered:

- 1. A composting plant accepting 50,000 tonnes of waste per year
- 2. A Mechanical Biological Treatment (MBT) plant accepting 50,000 tonnes of waste per year
- 3. An anaerobic digestion plant accepting 50,000 tonnes of waste per year
- 4. A pyrolysis / gasification plant accepting 50,000 tonnes of waste per year
- 5. A small landfill site accepting 25,000 tonnes of waste per year releasing emissions from a landfill gas flare, and fugitive emissions of uncaptured gas (75% of landfill gas assumed to be captured and burnt in the flare)
- A small landfill site accepting 25,000 tonnes of waste per year releasing emissions from a 1 MegaWatt (MW) landfill gas engine, and fugitive emissions of uncaptured gas (75% of landfill gas assumed to be captured and burnt in the engine)
- 7. A medium landfill site accepting 75,000 tonnes of waste per year releasing emissions from a landfill gas flare, and fugitive emissions of uncaptured gas (75% of landfill gas assumed to be captured and burnt in the flare)
- 8. A medium landfill site accepting 75,000 tonnes of waste per year releasing emissions from three 1 MW landfill gas engines, and fugitive emissions of uncaptured gas (75% of landfill gas assumed to be captured and burnt in the engines)
- 9. A large landfill site accepting 350,000 tonnes of waste per year releasing emissions from a landfill gas flare, and fugitive emissions of uncaptured gas (75% of landfill gas assumed to be captured and burnt in the flare)
- 10. A large landfill site accepting 350,000 tonnes of waste per year releasing emissions from eight 1 MW landfill gas engines, and fugitive emissions of uncaptured gas (75% of landfill gas assumed to be captured and burnt in the engines)

Volatile organic compounds are released as fugitive emissions from landfill sites. Unit risk factors have been specified by the WHO for benzene and chloroethene. These substances were included in the study. Volatile organic compounds are also released from combustion processes, but in much lower quantities. These emissions have therefore not been considered in this study.

Combustion gases result from many waste management processes, including the combustion of landfill gas in a flare or for the generation of electricity; combustion of syngas following pyrolysis/gasification; or combustion of biogas following

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

anaerobic digestion. The combustion gases contain products of combustion such as sulphur dioxide, nitrogen oxides and particulate matter.

Emissions of metals, dioxins and furans and PAHs may arise from combustion processes or from fugitive emissions of landfill gas. Dioxins and furans and PAHs can be formed during the combustion process itself, although steps are taken to minimise the formation of these substances.

The substances for which emissions estimates are available are set out in Table 4.1.

	Composting	МВТ	Anaerobic Digestion	Pyrolysis / Gasification	Incineration	Landfill
NO _x	-	Y	Υ	Υ	Y	Y
SO ₂	-	Y	Υ	Υ	Y	Y
PM ₁₀	N ¹	Y	Υ	Υ	Y	Y
PAHs	Ν	Ν	Ν	Ν	Y	N
Dioxins & Furans	Ν	Ν	(Y)	(Y)	(Y)	(Y)
Benzene	Ν	Ν	-	-	-	Y
Vinyl Chloride	Ν	Ν	-	-	-	Y
Arsenic	Ν	Ν	Υ	Υ	Y	Y
Chromium VI	Ν	Ν	Ν	Ν	Y	N
Nickel	Ν	Ν	Υ	Υ	Y	Y
Micro-organisms	Ν	Ν	-	-	-	N

Table 4.1 Substances assessed

Notes:

Y assessed – quantitative forecast

(Y) assumed to be insignificant, by reference to Environment Agency (2003l; draft)

N not assessed, potentially significant emission

- not assessed because no significant emission

Note 1: Given the uncertainty in measured emissions to air from composting, a calculation of the health burden of emissions has not been carried out (see Section 4.9.4)

The summary in Table 4.1 above highlights the following areas where data are available, and where data are lacking:

Incineration:	A wide range of emissions data is available
Landfill:	A wide range of emissions data is available, although emissions of micro-organisms may be significant, together with exposure via other routes.
Pyrolysis/gasification:	A range of emissions data is available, but this is based on a limited data set
Anaerobic digestion:	A range of emissions data is available, but this is based on a limited data set

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Composting:	Emissions data are in general not available. A low or nil forecast impact should not be taken to demonstrate no effect.
MBT:	Emissions data are in general not available. A low or nil forecast impact should not be taken to demonstrate no effect.
PAHs:	Little data are available. The assessment of incineration indicates that these emissions are likely to be insignificant
Micro-organisms:	No exposure-response functions are available for micro-organisms. There is no emissions data available, although some environmental measurements have been published. This is a potential issue for landfill, composting and MBT

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4.4 Process Data

Data for the emission rates of the various pollutants released to the atmosphere were taken from Chapter 2. Where data are unavailable a note has been made in the Notes section under the table.

Release conditions including area of landfill site and capacity, stack heights, stack internal diameters, efflux velocity, efflux temperature and volume flow rates, are given in Table 4.2.

There are many different sizes and operations of each process within each process type. Characteristic release conditions were therefore chosen for each process type, based on the information in Chapter 2, and our experience of waste management facilities.

The Environment Agency (2003l; draft) study found that modelled public exposure to dioxins and furans from illustrative municipal waste incinerator facilities was between 0.3% and 0.8% of the modelled background exposure. It was concluded that "the dioxin emission contribution to exposure of local populations is entirely negligible" (Chapter 7 of Environment Agency, 2003l; draft). Furthermore, there is no dose-response coefficient for dioxins based on human exposure data with no threshold for effects (the evidence suggests that it may be a non-genotoxic carcinogen, which would be characterised by a threshold for adverse health effects). On this basis, the health consequences of exposure to dioxins and furans were not investigated in this study.

Emissions from landfill processes differ from other processes, in terms of the timescale over which emissions arise following disposal to landfill. The majority of emissions of landfill gas and leachate from biodegradable waste materials typically take place over a period of some 20 years following disposal. However, generation of gas and leachate will continue at a lower rate for many years. This means that the landfill gas being emitted from landfilled MSW at present is due mostly to wastes deposited over the past 20 years. Conversely, the MSW deposited at present will degrade and release most of its landfill gas over a period of around 20 years. This means that the health consequences due to emissions from landfill set out below will be spread over a period of at least 20 years. At the same time, there will also be health consequences due to emissions from wastes deposited over the past 20 years.

If waste of similar composition were being deposited to landfill at a similar rate and being managed in a similar way, then emissions of landfill gas and any health consequences would reach a point where the instantaneous release would be the same rate as if all emissions took place instantaneously. In fact, this is not the case, and the release rate of landfill gas from the UK as a whole will not be exactly the same as would arise if all emissions took place simultaneously. The impact of the Landfill Directive and other initiatives will be that current and future emissions of landfill gas will be lower than in the past

Table 4.2 below shows the input data used for each process.

Introduction	Review of Review of Information on Epidemiological Research	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Table 4.2 Input data for quantification of health effects of emissions to air

	Unit	Com- posting	MBT	Anaerobic digestion	Pyrolysis gasification	Inciner -ator	Landfill – small + flare	Landfill – medium + flare	Landfill – large + flare	Landfill – small + engine	Landfill – medium + engine	Landfill – large + engine
Site Description												
Waste receipts	T/yr	50,000	50,000	50,000	50,000	122,64 0	25,000	75,000	350,000	25,000	75,000	350,000
Fugitive Emissions	6											
Surface area of surface emissions	m²	n/a	n/a	n/a	n/a	n/a	20,000	20,000	30,000	20,000	20,000	30,000
Height of release area	m	n/a	n/a	n/a	n/a	n/a	0	0	0	0	0	0
Flow rate (25% of gas generated)	m³/hr	n/a	n/a	n/a	n/a	n/a	143	428	1998	143	428	1998
Release temperature	°C	n/a	n/a	n/a	n/a	n/a	25	25	25	25	25	25
Receptor grid				Suffici	ent to give cond	entrations	< 0.01 µg/m3	3, starting fron	n 2 km × 2 km	; 25 m interval		
Emission rate of benzene	g/m2/s	n/a	n/a	n/a	n/a	n/a	9.5 × 10 ⁻⁹	2.9 × 10 ⁻⁸	8.9 × 10 ⁻⁸	$9.5 imes 10^{-9}$	$2.9 imes 10^{-8}$	$8.9 imes 10^{-8}$
Emission rate of chloroethene	g/m2/s	n/a	n/a	n/a	n/a	n/a	4.4 × 10 ⁻⁸	1.3 × 10 ⁻⁷	4.1 × 10 ⁻⁷	4.4×10^{-8}	$1.3 imes 10^{-7}$	4.1×10^{-7}
Point Emissions												
Number of release points/flues		1	1	2	2	1	1	1	1	1	3	8
Diameter of each release point	m	0.5	0.5	1	1	1.77	1.0	1.4	2.2	0.3	0.3	0.3



Introduction	Review of Information on Emissions	Review of Review of Information on Epidemiological Emissions Research		Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions	
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities	

 Table 4.2
 Input data for quantification of health effects of emissions to air (continued)

	Unit	Com- posting	МВТ	Anaerobic digestion	Pyrolysis gasification	Inciner- ation	Landfill – small + flare	Landfill – medium + flare	Landfill – large + flare	Landfill – small + engine	Landfill – medium + engine	Landfill – large + engine
Height of release point	m	15	15	30	30	76	5	8	15	6.5	6.5	6.5
LFG combustion rate (75% of gas generated)	m³/hr	n/a	n/a	n/a	n/a	n/a	428	1284	5993	428	1284	5993
Release velocity	m/s	15	15	15	15	23.78	2.5	4.0	7.5	29.7	29.2	51.5
Release temperature	°C	40	150	150	200	161	1000	1000	1000	450	450	450
Receptor grid				Suffic	ient to give con	centrations <	< 0.01 µg/m3, sta	arting from 2 k	m × 2 km; 25 m	n interval		
Emission rate of NOx	g/s		0.115	0.149	1.237	4.95	0.079	0.238	1.110	0.713	0.713	1.249
Emission rate of SO2	g/s		0.044	0.0024	0.082	0.872	0.063	0.190	0.888	0.055	0.055	0.097
Emission rate of PM10	g/s				0.019	0.111	0.006	0.019	0.089	0.0055	0.0055	0.0097
Emission rate of benz[a]pyrene	g/s	Not available	Not available	Not available	Not available	3.2 × 10 ⁻⁶	Not available	Not available	Not available	Not available	Not available	Not available
Emission rate of arsenic	g/s	0.0028	Not available	$4.0 imes 10^{-7}$	$9.5 imes10^{-5}$	0.00309	Not available	Not available	Not available	1.3 × 10 ⁻⁶	$1.3 imes 10^{-6}$	$2.2 imes 10^{-6}$
Emission rate of chromium	g/s	Not available	Not available	Not available	Not available	0.000643	Not available	Not available	Not available	Not available	Not available	Not available
Emission rate of nickel	g/s	0.011	Not available	2.4×10^{-7}	$6.3 imes 10^{-5}$	0.00140	Not available	Not available	Not available	1.0 × 10 ⁻⁵	1.0×10^{-5}	$1.8 imes 10^{-5}$

"Not emitted" implies no likelihood of being emitted at a rate comparable with other waste management facilities

Note: incineration emissions taken from *Environment Agency, 2003I; draft.* The emissions data for incinerators used in *Environment Agency, 2003I* differs from the data set out in Chapter 4.



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4.5 Dispersion modelling

4.5.1 Model used

The dispersion model ADMS 3.1 was used for this study. This is a mathematical model that is used to predict the concentration and deposition rates of atmospheric pollutants. ADMS 3.1 has been well validated against experimental studies of dispersion, both in field experiments and wind tunnel studies. ADMS 3.1 can be described as a new generation Gaussian model because the description of the atmosphere used by the model is based on up-to-date physics. It provides an improved description of the vertical turbulence structure of the atmosphere and, as a result, provides an improved description of pollutant dispersion. The model has some limitations, including the inability to model concentrations in cases with very low or zero wind speed and dispersion over very large distances (i.e. hundreds of km).

The accuracy of the results from this type of model depends on the quality of the data that describe the scenario. Validation studies indicate that atmospheric dispersion models provide a good approximation to ground-level concentrations at source-receptor distances of around 100 m to 1 kilometre, particularly for long-term averages.

4.5.2 Meteorological data

The meteorological data used in the *Environment Agency, (2003l; draft)* study was also used for this research. The data set used was Elmdon (Birmingham Airport) 1996. The Environment Agency study assessed data from a number of meteorological stations. The Elmdon dataset was identified as providing a set of data likely to provide concentrations in the middle of the range of those experienced around England and Wales.

4.5.3 Roughness length

A single value for roughness length (a measure of land surface characteristics) was chosen for each case modelled based on the surrounding area of the process modelled.

4.5.4 Grid area

A receptor grid area extending 1 km from the source in each cardinal direction was found to cover a suitable range. The grid resolution was 25 metres (that is, concentrations were calculated at 25 metre intervals).

4.5.5 Population data

An initial examination of results in *Environment Agency (2003l; draft)* showed that the results were sensitive to the distribution of population around the source of pollution. One example using a real distribution of population, showed that considering the same source at locations only 15km apart resulted in total health effects that differed by a factor of three. In order to be able to compare results between source types without the effect being hidden by local population density variation, uniform population densities were used for the final analysis. This is the approach that has been adopted for this research project.



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Population density figures were taken from a DETR report (Todorovic et al., 2000) that classified a number of location types according to rural, rural/suburban, suburban, urban/suburban and urban categories as well as more specific sub-categories. The two population densities chosen from the sub-categories in the Birmingham University report have also been used here.

To represent urban population densities the 'Industrial towns' category from the suburban/urban section was chosen (population density 3784 people/km²). This was applied to gasification/pyrolysis, anaerobic digestion and MBT processes.

For rural areas the 'Town and country' category from the suburban/rural section was chosen with a population density of 566 people/km². This was applied to landfill and composting processes.

Although there are population densities with larger and smaller values, these are unlikely to apply over the whole area being considered and the values chosen are likely to provide a representative value.

Baseline health data

Baseline health rate data have been extracted from the research being carried out by Birmingham University for the Environment Agency. In the report, baseline health rate data for the UK were calculated from data on hospital admissions from the Department of Health for 2000/2001 (Department of Health, 2001) and mortality data from the Office of National Statistics for 2000 (ONS, 2000a), normalised to population data for England or England and Wales for mid-2000 (ONS, 2000b). The criteria used were taken from a DEFRA report into the economic effects of air quality which used the COMEAP methodology (DEFRA, 2001). The data are summarised in Table 3.

Table 4.3	Baseline health rate data
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Description	Annual Baseline health rate (per 100,000)
Respiratory hospital admissions (rha)	846.4
Cardiovascular hospital admissions	690.5
Deaths from all causes	938.6

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4.6 Results

The predicted long term average concentrations were assessed in terms of their health consequences: number of deaths brought forward per year; increases in number of respiratory cases per year and increases in cardiovascular cases per year.

The estimated health consequences are set out in Table 4.6 and 4.7 below. This information is appropriate to typical facilities which may be operated in the UK. Any assessment of an individual facility would need to take account of the local circumstances and any known sensitivity to the emissions studied in this chapter.

Introduc	ction	Review of Information on Emissions	Review of Review of Information on Epidemiological Emissions Research		Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope		Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

 Table 4.4
 Estimated health impacts due to emissions to air (per facility per year)

	Substance	Composting	MBT	Anaerobic digestion	Pyrolysis Gasification	Incineration	Landfill – small + flare	Landfill – medium + flare	Landfill – large + flare	Landfill – small + engine	Landfill – medium + engine	Landfill – large + engine
Deaths	Total	No data	0.00091	0.000074	0.00154	0.0079	0.00062	0.0011	0.0017	0.00050	0.00087	0.00102
brought	PM ₁₀	No data	No data	No data	0.000347	<0.00005	0.00007	0.00012	0.00019	0.00006	0.00010	0.00011
forward	SO ₂	Not emitted	0.00091	0.000074	0.001192	0.0079	0.00055	0.00098	0.00154	0.00044	0.00077	0.00090
Pospiratory	Pedigree	n/a	Poor(3)	Moderate (5)	Moderate (6)	Moderate (6)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)
Respiratory	Total	No data	0.0025	0.0036	0.0147	0.192	0.00100	0.0018	0.0028	0.0047	0.0082	0.0096
admissions	PM ₁₀	No data	No data	No data	0.00033	<0.00005	0.00006	0.00012	0.00019	0.00005	0.00009	0.00011
	SO ₂	Not emitted	0.00068	0.00005	0.00090	0.0057	0.00042	0.00073	0.00116	0.00033	0.00058	0.00068
	NOx	Not emitted	0.00179	0.0035	0.0134	0.186	0.00052	0.00092	0.00144	0.00431	0.00752	0.00876
	Pedigree	n/a	Poor(3)	Moderate (5)	Moderate (6)	Moderate (6)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)
Cardio-	Total (PM ₁₀)	No data	No data	No data	0.00027	<0.00005	0.00005	0.00010	0.00015	0.00004	0.00008	0.000089
vascular admissions	Pedigree	n/a	n/a	n/a	Moderate (6)	Moderate (6)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)
Additional	Total	No data	No data	$5.4 imes 10^{-8}$	9.42×10^{-7}	<3.0 × 10 ⁻⁶	$1.19 imes 10^{-6}$	$3.57 imes 10^{-6}$	$1.68 imes 10^{-5}$	1.31×10^{-6}	$3.78 imes 10^{-6}$	1.70×10^{-5}
cancer	Arsenic	No data	No data	4.7×10^{-8}	8.05×10^{-7}	<7.0 × 10 ⁻⁷	No data	No data	No data	3.98×10^{-8}	$6.94 imes 10^{-8}$	7.81×10^{-8}
cases	Chromium (VI)	No data	No data	No data	No data	<7.0 × 10 ⁻⁷	No data	No data	No data	No data	No data	No data
	Nickel	No data	No data	7.1 × 10 ⁻⁹	1.36×10^{-7}	<7.0 × 10 ⁻⁷	No data	No data	No data	7.75×10^{-8}	1.35×10^{-7}	1.62×10^{-7}
	B [a] P	No data	No data	No data	No data	<7.0 × 10 ⁻⁷	No data	No data	No data	No data	No data	No data
	Vinyl chloride	No data	Not emitted	Not emitted	Not emitted	Not emitted	5.16 × 10 ⁻⁷	$1.55 imes 10^{-6}$	$7.27 imes 10^{-6}$	5.16 × 10 ⁻⁷	$1.55 imes 10^{-6}$	$7.27 imes 10^{-6}$
	Benzene	No data	Not emitted	Not emitted	Not emitted	Not emitted	6.75×10^{-7}	2.02×10^{-6}	9.51×10^{-6}	6.75×10^{-7}	2.02×10^{-6}	9.51 × 10 ⁻⁶
	Pedigree	n/a	n/a	Moderate (5)	Moderate (6)	Moderate (6)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)

Note: the health effects of landfill emissions will take place over a number of years. "Not emitted" implies no likelihood of being emitted at a rate comparable with other waste management facilities. The key areas of uncertainty are as follows:

- Uncertainty associated with emissions estimates: Lower for combustion parameters; higher for fugitive emissions. Lower for incineration and landfill; higher for other waste management facilities. Range between a factor of 1.1 and a factor of 5
- Uncertainty associated with COMEAP/WHO factors: A factor of 10 used in all cases, common to all forecasts. WHO estimates are more likely to be associated with systematic uncertainty because they are based on extrapolation from occupational exposures
- Uncertainty associated with air modelling methodology: approximately a factor of three



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

• Uncertainty associated with impacts which cannot be quantified: lower for incineration and landfill; higher for composting and MBT. Will tend to increase the estimated health impacts associated with these activities.

Table 4.5 Estimated health impacts due to emissions to air (per tonne of waste processed)

Health impact	Composting	МВТ	Anaerobic digestion	Pyrolysis gasification	Inciner- ation	Landfill – small + flare	Landfill – medium + flare	Landfill – large + flare	Landfill – small + engine	Landfill – medium + engine	Landfill – large + engine
Deaths brought forward Per tonne of waste	No data	1.82×10^{-8}	1.48 × 10 ⁻⁹	3.08 × 10 ⁻⁸	6.4 × 10 ⁻⁸	2.47×10^{-8}	1.5 × 10 ⁻⁸	4.9 × 10 ⁻⁹	2.0 × 10 ⁻⁸	1.2 × 10 ⁻⁸	2.9 × 10 ⁻⁹
Respiratory admissions Per tonne of waste	No data	$4.95 imes 10^{-8}$	$7.20 imes 10^{-8}$	2.93 × 10 ⁻⁷	1.5 × 10 ⁻⁶	4.00×10^{-8}	2.4 × 10 ⁻⁸	8.0 × 10 ⁻⁹	1.9 × 10 ⁻⁷	1.1 × 10 ⁻⁷	2.7 × 10 ⁻⁸
Cardio-vascular admissions Per tonne of waste	No data	No data	No data	5.45 × 10 ⁻⁹	<4.1 × 10 ⁻¹⁰	2.07 × 10 ⁻⁹	1.3 × 10 ⁻⁹	4.3 × 10 ⁻¹⁰	1.7 × 10 ⁻⁹	1.0 × 10 ⁻⁹	2.5×10^{-10}
Additional cancer cases Per tonne of waste	No data	No data	1.08×10^{-12}	1.9×10^{-11}	<2.0 × 10 ⁻¹¹	4.8×10^{-11}	4.8×10^{-11}	4.8 × 10 ⁻¹¹	5.2 × 10 ⁻¹¹	5.0 × 10 ⁻¹¹	4.9 × 10 ⁻¹¹
Pedigree	n/a	Poor(3)	Moderate (5)	Moderate (6)	Moderate (6)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)	Poor (4)

Note: the health effects of landfill emissions will take place over a number of years. The key areas of uncertainty associated with these estimates are as follows:

- Uncertainty associated with emissions estimates: Lower for combustion parameters; higher for fugitive emissions. Lower for incineration and landfill; higher for other waste management facilities. Range between a factor of 1.5 and a factor of 10
- Uncertainty associated with COMEAP/WHO factors: A factor of 10 used in all cases, common to all forecasts. WHO estimates are more likely to be associated with systematic uncertainty because they are based on extrapolation from occupational exposures
- Uncertainty associated with air modelling methodology: approximately a factor of 10

Overall uncertainty: factor of between 14 and 20, obtained by combining above three components of uncertainty

• Uncertainty associated with impacts which cannot be quantified: lower for incineration and landfill; higher for composting and MBT. This area of uncertainty will tend to increase the estimated health impacts associated with these activities.



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

4.7 Comparison of health effects of emissions to air from waste management facilities

The quantification of health effects addresses only the effects of emissions to air of a subset of pollutants. It does not therefore provide a complete comparison between waste management facilities. This is firstly because emissions data is more complete for some facilities than others. Secondly, some processes are likely to emit greater quantities than other processes of substances which cannot be assessed in this way (e.g. bioaerosols). Thirdly, health impacts via pathways other than exposure to emissions to air are likely to be significant to varying degrees between the different waste management facilities. Emissions from landfill to groundwater could potentially constitute a significant pathway for adverse health effects, although it is difficult to propose a general relationship between health impacts of groundwater consumption and proximity to waste management facilities. Since 1988, for all major facilities, a site specific risk assessment has been carried out to ensure that any environmental impacts are limited to a low and acceptable level.

The data quality should also be borne in mind when interpreting the forecasts set out in Tables 4.4 and 4.5. The emissions data have associated uncertainty ranges, as set out in Chapter 2. The pedigree of the emissions data is also described in Chapter 2. The dispersion modelling component introduces additional uncertainties into the forecasts of health consequences, and has the effect of reducing the data pedigree to some extent. Similarly, the factors used to quantify the health consequences of exposure to emissions to air have an associated uncertainty, as set out in Chapter 2.

The assessment of emissions to air relates primarily to typical emissions under normal operating conditions. From time to time, problems arise at waste management facilities which could result in increases in emissions for short periods: these are not explicitly considered in this quantification of the health consequences of emissions to air. The most likely abnormal operating conditions are described in Chapter 2. Any such short-term increases in emissions are not likely to have a significant effect on long-term mean exposures.

The assessment is carried out for typical facilities assuming study inputs which are representative of the UK as a whole. This makes the results useful for a study on a national basis, but these study inputs may not be appropriate for specific facilities. For example, in areas of higher or lower population densities than those assumed in the study, the number of deaths brought forward or hospital admissions would be expected to be proportionately more or less significant. Similarly, in areas where the base rate of hospital admissions or deaths differ from the national average, the health effects would be expected to be proportionately more or less significant. In areas where the meteorology differs significantly from the Birmingham Airport dataset used in this study, the concentrations resulting from emissions from waste management facilities would differ. This would affect the forecast health impacts, although this is likely to be a relatively small influence, and is taken into account in the uncertainty assigned to the model forecasts. Finally, individual sites may have different emissions control regimes which would affect the applicability of the forecast emissions estimates. This is again taken into account in the overall uncertainty estimates and Pedigree evaluation for the emissions data.

Bearing these constraints in mind, the following conclusions can be drawn:

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

- The more significant effects are associated with emissions of the "classical" pollutants particulate matter (PM₁₀), sulphur dioxide and oxides of nitrogen. Emissions of carcinogens such as arsenic and nickel are less significant (Confidence level for this conclusion: Moderate).
- The effect of emissions from waste management on cardiovascular admissions is less significant than the effect on respiratory hospital admissions and deaths brought forward (Confidence level for this conclusion: Moderate).
- The most likely outcome identified in this study is approximately one additional respiratory hospital admission in five years as a result of emissions from an individual waste incinerator facility. On a national scale, this would currently correspond to approximately four respiratory hospital admissions per year. This level of effect would not be detectable by any practicable means. This forecast is reliable to within a factor of 30, with a moderate level of confidence.
- One death might be expected to be brought forward due to emissions from an individual incinerator approximately every 100 years. This forecast is reliable to within a factor of 30, with a moderate level of confidence. For other waste management facilities, one death brought forward might be expected to occur every 1000 years. For landfill, this forecast is reliable to within a factor of 30, but with a poor level of confidence. (*Environment Agency, 2003I; draft*).
- Impacts per tonne of waste for landfill are forecast to be lower as the size of the facility increases. Again, it should be borne in mind that the health effects of landfill emissions will take place over a number of years. By the same token, any health effects taking place at present will be as a result of wastes landfilled over previous years.
- For the cancer outcomes assessed, the incremental risks of leukaemia and haemangiosarcoma were at a similar level, and more significant than the lung cancer outcomes (Confidence level for this conclusion: Moderate).
- The potential health effects of emissions to air from composting cannot be assessed because there are no emissions data. Of the substances assessed, composting facilities are most likely to emit significant quantities of particulates. This represents a key area of uncertainty, and future research should be focused in this area.

In order to assess the potential health effects of any individual facility, it would be necessary to consider the local conditions and sensitivity to air pollution – for example, the density of residential properties in the area around the waste management facility.

A comparison of the key health outcomes (deaths brought forward; respiratory hospital admissions; cancer) between the waste management facilities is set out in Figures 4.1 to 4.3. For landfill, the median value for small, medium and large facilities was used.
Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Figure 4.1 Estimated deaths brought forward per tonne of waste processed



Figure 4.2 Estimated respiratory hospital admissions per tonne of waste processed



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Figure 4.3 Estimated additional cancer cases per tonne of waste processed



Figure 4.4 shows a comparison of deaths brought forward associated with the individual landfill scenarios considered:



Figure 4.4 Estimated deaths brought forward per tonne of waste processed (landfill scenarios)

210

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Figure 4.5 shows a comparison of deaths brought forward associated with each waste management option, but with uncertainties common to each option removed. The uncertainties associated with the dispersion modelling and use of the COMEAP/WHO factors have been removed from the estimates. The uncertainties shown in Figure 4.5 reflect only the uncertainties associated with the estimated emissions per tonne of waste processed. This figure can be used to identify the extent to which the effects associated with each waste management option differ.

Figure 4.5 Estimated deaths brought forward per tonne of waste processed (uncertainties common to each option removed: uncertainties associated with emissions per tonne of waste retained)



Note 1: this figure is provided to identify the extent to which the effects associated with each waste management option differ. Data Pedigree descriptions are those for the emissions estimates only, not for the combined emissions/dispersion modelling/health consequence evaluation. The actual uncertainties are shown in Figure 4.1.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Scope	Method	Processes and Substances considered	Process Data	Dispersion Modelling	Results	Comparison between Waste Management Facilities

Figure 4.6 shows the estimated additional health consequences based on the quantities of waste processed in 2000/2001. These outcomes arise from emissions to air from landfill and incineration only. Although some composting of MSW was also carried out in 2000/2001, there are no air emissions estimates.

Figure 4.6 Estimated annual UK health consequences due to waste processed in 2000/2001



Introduction	Review of Information on Emissions Review of Epidemiological Research		Review of Epidemiological Research	Quantification of Health Consequences of Emissions		Quantification of the Environmental Consequences of Emissions	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Introduction		Sourc	es of information		Main findings			Shortcomings in Inform	nation

5. QUANTIFICATION OF THE ENVIRONMENTAL CONSEQUENCES OF EMISSIONS

Introduction	Review of Information Emissions	on Review of Epidemiological Research	view of Quantification of Health idemiological Consequences of search Emissions		Quantification of the Con Environmental Hea Consequences of Rev Emissions Performance		ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Introduction		Sources of information		Main findings			Shortcomings in Inform	nation

Summary – environmental consequences of emissions

We carried out a detailed literature search to investigate whether there is evidence for environmental effects due to waste management operations. In fact, we found relatively little published evidence on environmental effects of treating and disposing of municipal solid waste (MSW).

We found that emissions of greenhouse gases were the most significant reported environmental effect – mainly arising from landfill of MSW. These emissions can be substantially reduced by collecting and burning the gas generated from decomposition of MSW in landfills.

We found that odours and effects on air quality were identified as issues which may be significant for a range of waste management process, unless they are properly controlled. Landfill and incineration were the processes for which the most significant impacts are reported in the literature.

We identified a number of areas where more detailed research would be useful. These include studies of the pathways by which materials released from waste management pass through and accumulate in ecosystems; considering the effects that these emissions might have on sensitive species; and calculating the significance of emissions to air of greenhouse gases.

Introduction	Review of Information Emissions	n on Review of Epidemiological Research	Quantificatio Consequence Emissions	n of Health es of	Quantification of the Environmental Consequences of Emissions	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Introduction Sou		Sources of information		Main findings			Shortcomings in Inform	nation

5.1 Introduction

This chapter draws together the available literature evidence for the existence or otherwise of environmental effects due to waste management operations. This chapter does not contain comments on environmental effects which are not supported by the evidence under review.

Waste management operations have a number of potential environmental impacts, the majority of which could occur only in the area surrounding the operation. Others occur over greater distances, either as a consequence of transfer of emissions through the air or water, or by the disposal of residues to a specific location.

An extensive search has been undertaken to identify the available literature, as described later. However, the search identified that in comparison to the data available investigating the potential health impacts of waste management operations, there is only a limited amount of data available on the environmental consequences of waste management.

Due to the limited numbers of information sources the data quality is somewhat lower than that available for health impacts. However, the information that is available has been assessed in detail. This chapter summarises the search strategy, the data sources, the environmental effects identified are summarised in table 5.1, and a discussion on the shortcomings in the data reviewed. An element of judgement has been used to derive this summary table. Appendix 2 contains a detailed description and review of the information subsequently summarised in this chapter.

When considering the environmental effects of an individual facility, it is important to take account of the local environmental conditions, and any evidence of sensitivity to the potential environmental effects of the waste management facility.

The bibliography for this chapter is provided in Appendix 1 to the report.

Introduction	Review of Information Emissions	n on Revi Epid Rese	view of idemiological search	Quantification Consequence Emissions	n of Health es of	Quantification of the Environmental Consequences of Emissions	Conte Healt Envire Revie Perce	ext for Quantified h and onmental Risks, & ev of Public eption Issues	Conclusions
Introduction Sou		Sources of inf	information		Main findings			Shortcomings in Inform	nation

5.2 Sources of Information

Three strategies were adopted to identify relevant research on the environmental effects of waste management.

- A review was carried out of environmental statements held by the Institute for Environmental Management and Assessment (IEMA). The IEMA library contains approximately 50 environmental statements prepared in respect of waste management facilities. Eight environmental statements were selected so as to allow the range of facilities addressed in this project to be considered. These included integrated facilities incorporating a range of waste management processes.
- A search of published literature using the British Library's facilities. The search strategy adopted is set out below.
- Knowledge of relevant research within the project team.

5.2.1 Published literature search strategy

Searches were carried out over the following databases:

- Biological Abstracts(1980–2003)
- BIOSIS Previews (1970–2003) (Life sciences abstracting service)
- CAB Abstracts (1973–2003) (bibliographic database covering agriculture, forestry, aspects of human health, human nutrition, animal health and the management and conservation of natural resources)
- Cambridge Scientific Abstracts Environmental Science (1984–2003) (Multidisciplinary database covering all aspects of environmental science)
- Current Contents Agricultural, Biological, and Environmental Sciences (2000– 2003)
- SIGLE (1980–2003) (Bibliographic database covering European non-conventional ("grey") literature in the fields of pure and applied natural sciences and technology, economics, social sciences, and humanities)
- Science Direct
- SciFinder

These databases were searched following the following procedure:

Step 1: A search was carried out for the following terms individually in title, abstract or subject:

Landfill; recycling; incineration; composting; gasification; pyrolysis; energy from waste; anaerobic digestion; mechanical biological treatment; waste transfer/transport



Introduction	Review of Informatio Emissions	on on	Review of Epidemiological Research	Quantificatio Consequenc Emissions	n of Health es of	Quantification of the Environmental Consequences of Emissions	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Introduction Sour		Sourc	es of information		Main findings			Shortcomings in Inform	nation

Step 2 : A search was carried out for **refuse; municipal solid waste** in title, abstract or subject: adjacent to one of the following terms: **recycle; treatment; transfer; disposal**

Step 3 : A combined list of abstracts from Step 1 and 2 was produced. [Research relating to waste]

Step 4: A search was carried out for **environmental** in title, abstract or subject: adjacent to one of the following terms: **impact; effect; risk; hazard; contamination**

[research relating to environmental effects]

Step 5: A search was carried out for each the following terms individually in title, abstract or subject: **Assessment; evaluation; measurement**

Step 6 : A combined list of abstracts identified in both Step 4 and Step 5 was produced.

[research relating to assessment/measurement of environmental effects]

Step 7 : A combined list of abstracts identified in both Step 3 and Step 6 was produced.

[research linking assessment of environmental effects to waste]

Step 8 : The list of abstracts identified at Step 7 was limited to English language [Final abstracts database: research in English linking assessment of environmental effects to waste]

This procedure produced a total of 1500 titles. These were reviewed to identify potentially relevant abstracts. Approximately 500 abstracts were reviewed, and from this a total of 300 papers, books and other publications were reviewed. These are contained in the Bibliography (Appendix 1 to this report).

Each type of facility is considered in detail against each impact in appendix 2. A summary is then provided in Table 5.1.

Introduction	Review of Information Emissions	on Review of Epidemiological Research	Quantificatio Consequenc Emissions	n of Health es of	Quantification of the Environmental Consequences of Emissions	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Introduction Sou		Sources of information		Main findings			Shortcomings in Inform	nation

5.3 Main findings

A summary of the effects identified is given in Table 5.1. This table is derived from the analysis detailed in Appendix 2. Based on this summary, Table 5.2 identifies the key environmental issues associated with waste management operations.

The most significant environmental impact was identified as emissions of greenhouse gases from landfill of MSW. This is an issue which affects all landfill sites accepting MSW, and the contribution of methane emitted from landfills to global warming is significant. A considerable benefit can be achieved by collection and combustion of landfill gas, but there is a practical limit to the proportion of the gas generated which can be collected in this way. Because of the significance of methane emissions from landfill, alternatives to landfill for MSW are often viewed as having a positive benefit on global warming by reducing the need to landfill biodegradable waste.

While visual intrusion might be expected to be a significant issue for most or all kinds of waste management facilities, we did not find literature evidence to support this.

In general, odours and effects on air quality were identified as issues which may be significant for a range of waste management process, unless properly controlled. Landfill and incineration were the processes for which the most significant impacts are reported in the literature.

	Emissions	Epidemilological Research	Emissions		Consequences of Emissions		eption Issues	Conclusions
Introduction	Review of Information	n on Epidemiological	Quantificatio	on of Health	Quantification of the Environmental		ext for Quantified h and comontal Risks &	Conclusions

Table 5.1 Summary of observed environmental effects reported in the literature review

Activity	Noise	Odour	Dust	Flora/fauna	Soils	Water quality/flow	Air quality ¹	Climate	Building damage (acid gases)
Composting	+7 to +32 dBA increment recorded. Complaints recorded at up to 150 m, but relatively few	Contribute up to 680 OU/m ³ . Complaints up to 80 m	Typical of industrial area – adjacent to site only	Low risk of effect due to emissions to water: indirect benefit from replacement of peat with compost	Benefit from improving soil structure. Potential effect of elevated levels of metals	Possible effect due to emissions to water/sewer, but water normally recirculated within process	Micro-organisms up to 10 ⁶ cfu/m ³ . May be significant at distance of up to 250 m. Can be abated with in-vessel system	Small effect due to CO ₂ and possibly emissions, but less than landfill	Nil
Mechanical biological treatment	Noise from shredding could be significant, but can normally be controlled with proper mitigation	Could be significant: abatement would normally be required	Might be significant, although dust generating activities carried out indoors	Low	Low	Possible effect due to rainwater or washdown water runoff	Micro-organisms and VOCs may be significant. Would normally be abated	Small effect due to CO ₂ emissions, but less than landfill	Low or Nil
Materials recycling facility	Low potential. +15 to +20 dBA increment recorded. No complaints normally	Nil at clean MRF, no complaints normally. Low potential for complaints at dirty MRF	Low potential. No complaints normally	Low	Low	Possible effect due to rainwater or washdown water runoff at dirty MRF	Micro-organisms up to 5000 cfu/m ³ , unlikely to be significant	Slight overall benefit	Nil
Anaerobic digestion	Could be significant, but can normally be controlled with proper mitigation	No complaints normally. Odours from waste storage/ processing can be controlled via combustion air	Low potential. No complaints normally	Low risk of effect due to emissions to water: may be indirect benefit from replacement of peat with composted end product	May be benefit from improving soil structure from use of composted end product. Potential effect of elevated levels of metals	Possible effect due to elevated N levels in wastewater (290 – 500 litres per tonne)	Emissions of combustion products. Offset to some extent by avoided power generation emissions	Small effect due to CO ₂ emissions, but less than landfill	Minor adverse effect



Introduction	Review of Information o Emissions	n Review of Epidemiological Research	Quantificatio Consequence Emissions	on of Health ses of	Quantification of the Environmental Consequences of Emissions	Conte Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Introduction	s	Sources of information		Main findings			Shortcomings in Inform	nation

Activity	Noise	Odour	Dust	Flora/fauna	Soils	Water quality/flow	Air quality ¹	Climate	Building damage (acid gases)
Gasification/ pyrolysis	+5 dBA increment recorded. Some noise sources could be significant, but can normally be controlled with proper mitigation	Potentially significant, but odours from waste storage/ processing are normally controlled via combustion air	Can normally be adequately controlled via combustion air. Ash handling/ transportation is possible source of dust	Low	Low	Low	Minor contribution to local levels of NOx (likely to be less than incineration)	Small effect due to CO ₂ emissions, but less than landfill	Minor adverse effect
Unsegregated Incineration	Some noise sources could be significant, but can normally be controlled with proper mitigation	Potentially significant, but odours from waste storage/ processing are normally controlled via combustion air	Can normally be adequately controlled via combustion air. Ash handling/ transportation is possible source of dust	Potentially significant risk of accumulation of metals and dioxins and furans, though other sources more significant. No adverse animal health effects observed	Potentially significant risk of accumulation of metals and dioxins and furans, though other sources more significant and found not to be significant health issue. Dioxins: Contribution 0.1 – 1 ng/kg compared to background of 0.1 – 100 ng/kg	Potentially significant risk of contaminants leaching from ash. Contributes less than 20% of contaminants in precipitation	Minor contribution to local levels of NOx and metals (contribution typically 1 – 10% of local background)	Small effect due to CO ₂ emissions, but less than landfill	Minor adverse effect
Small-scale Incineration with pre- sorting	Some noise sources could be significant, but can normally be controlled with proper mitigation	Potentially significant, but odours from waste storage/ processing are normally controlled via combustion air	Can normally be adequately controlled via combustion air. Ash handling/ transportation is possible source of dust	Low – risk of accumulation of dioxins and furans minimised because pre- sorting removes precursors	Low – risk due to deposition of dioxins/ furans and metals minimised because pre-sorting removes precursors	Low – risk of contaminants leaching from ash minimised because pre- sorting removes precursors	Minor contribution to local levels of NOx and metals.	Small effect due to CO ₂ emissions, but less than landfill	Minor adverse effect
Landfill	5% of complaints caused by noise. +5 to +10 dBA increment recorded	60% of complaints caused by odour. Complaints recorded at up	1% of complaints caused by dust. Complaints recorded at up to 250 m	Potential effect in the event of leachate escaping or from vermin. Restoration may	Potentially significant as large area may be excavated	Potentially significant, but generally low/non detectable effects from	Potentially significant effects from engine/flare emissions (NOx, metals) or VOCs	Significant adverse effect mainly due to methane emissions, even if landfill	Minor adverse effect



Introduction	Review of Information Emissions	on on	Review of Epidemiological Research	Quantification Consequence Emissions	on of Health ses of	Quantification of the Environmental Consequences of Emissions	Cont Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
Introduction Sour		Sourc	rces of information		Main findings			Shortcomings in Inform	nation

Activity	Noise	Odour	Dust	Flora/fauna	Soils	Water quality/flow	Air quality ¹	Climate	Building damage (acid gases)
		to 2 km		provide improved habitat		current UK landfills	from fugitive gas	gas collected and burnt	
Transportation / Transfer stations	Low potential for noise nuisance	Odours could potentially be significant. Normally controlled by minimising turnaround time	Low	Low	Low	Possible effect due to rainwater or washdown water runoff	Low. Potentially minor impact from micro- organisms	Minor benefit due to more efficient logistics	Nil

Note:

1: The air quality assessment does not include the specific impacts of vehicle emissions as these are normally insignificant in the local context of individual MSW proposals compared with background traffic emissions.



	Introduction	Review of Information Emissions	n on	Review of Epidemiological Research	Quantificatio Consequenc Emissions	n of Health es of	Quantification of the Environmental Consequences of Emissions	Cont Healt Envir Revie Perce	ext for Quantified h and onmental Risks, & ew of Public eption Issues	Conclusions
I	Introduction		Sourc	es of information		Main findings			Shortcomings in Inform	nation

 Table 5.2
 Summary of key environmental issues reported in the literature review

Activity	Noise	Odour	Dust	Flora/fauna	Soils	Water quality/flow	Air quality	Climate	Building damage
Materials recycling facility	×	×	×	×	×	××	* *	-	-
Composting	* *	* * *	××	✓	×	**	* * *	×	-
Mechanical biological treatment	* *	* * *	××	-	-	* *	* *	×	×
Anaerobic digestion	* *	* *	×	×	×	**	* *	×	×
Gasification/ pyrolysis	* *	* *	××	-	-	-	* *	×	×
Incineration with pre- sorting	* *	* *	* * *	* *	××	××	* * *	×	×
Incineration	**	**	***	***	***	***	***	×	×
Landfill	* * *	* * *	××	*** 🗸	* * *	* * *	* * *	****	×
Waste transfer stations	* *	* * *	×	-	-	××	×	· · · · · · · · · · · · · · · · · · ·	-

Category	Meaning
✓	Direct or indirect benefit
-	No effect
×	Unlikely to be significant
××	Potentially significant impact in some cases, but can be controlled
* * *	Impact can normally be controlled, but an issue at sites where design, engineering or operation falls below best practice
****	An issue at all sites

Note: These issues are identified from historical performance and do not take account of current and future changes in regulation and operation

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of H Consequences of Emissions	lealth	Quantification of the Environmental Consequences of Emissions	Context for Quan Health and Environmental Ri Review of Public Perception Issues	tified isks, & s	Conclusions
Introduction	-	Sources of information		Main f	indings		Shortco	omings in Information

5.4 Shortcomings in information

Whilst undertaking the literature search, it has been noted that there is currently only a limited amount of information available on the potential environmental effects of waste management operations. The environmental effects of waste management are less investigated than the potential health impacts for a number of reasons:

- The environmental effects are less in the public forum than potential health impacts;
- Landfills are seen as the norm in the UK, and consequently have not attracted scientific investigation of potential environmental effects;
- The regulatory requirements for environmental assessment of the potential effects of waste management operations, has until recently, been less than that required for health impacts;
- Recent investigations surrounding the health impacts of waste management facilities have tended to overshadow any investigations of environmental impacts.

As a consequence there are only limited data available relating to the potential environmental effects of waste management operations. There are several areas which have been identified during the course of this investigation where data are currently lacking; these include:

- Assessment of the potential for bioaccumulation and biomagnification of released materials in flora, fauna, aquatic environments and soils;
- Assessment of chronic effects on flora and fauna of exposure to released materials;
- Quantification of the effects on global climate of emissions of carbon dioxide, methane and other 'greenhouse' gases;
- Quantification of acute and chronic contamination of surface and groundwater as a consequence of planned and unplanned releases;
- Assessing the accumulation of metals, hydrocarbons and other contaminants associated with the spreading of MSW derived compost;
- Assessment of the impacts of emissions on habitats and biodiversity, and the potential for loss of species;
- Assessment of the likely effect of changes in facility design.

As a result of these shortcomings in the research, there are many unknowns when assessing the potential environmental impacts of acute and chronic emissions associated with waste management options. Very little data exists to be able to quantify the point at which emission concentrations become harmful or of the ultimate environmental effects of released contaminants.

As has been discussed previously, emissions data exist for most waste management operations. However, there are little field data on the fate of the emissions and the pathways by which emissions will disperse in the environment.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction	ı	Comp	parison between MSW manage	ement options	Comparison between MSW management ar	id other activities

6. CONTEXT FOR QUANTIFIED HEALTH AND ENVIRONMENTAL RISKS, AND REVIEW OF PUBLIC PERCEPTION ISSUES

Introduction	troduction Review of Review of Review of Information on Er Emissions Review of Review		view of Quantification of Health idemiological Consequences of Emissions			Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction	1	Comp	arison between MSW manage	ment options	Com	nparison between MSW management an	d other activities

Summary – contextual information and public perception

This chapter sets the estimated health and environmental effects of municipal solid waste (MSW) treatment and disposal in context.

We have considered emissions from MSW treatment and disposal alongside other widespread activities in the UK. This shows that managing MSW accounts for less than 2.5% (one fortieth) of almost all emissions which we were able to quantify. The exceptions to this are emissions to air of methane (MSW accounted for nearly 30% of the UK total) and cadmium (10% of the national total, of which most arises from landfill). Some of the information on emissions to air from waste management facilities was of poor quality, and could helpfully be improved.

A similar picture emerges when considering the health effects of emissions to air from facilities treating and disposing of MSW. We reviewed the estimated health effects of MSW against other comparable causes of ill-health, comprising skin cancer, lung cancer from passive smoking, and air pollution in general. We also looked at other health hazards like accidents in the home or workplace, accidents caused by fireworks, traffic accidents or environmental factors such as excessive cold. Even though some of the information was of poor quality, we found that other influences on health appear to be much more important than waste management. This is true even for people living very close to municipal solid waste management facilities.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction	Introduction		arison between MSW manage	ment options	Comparison between MSW management an	d other activities

6.1 Introduction

Waste management is one of many activities undertaken today which may impact on human health and the environment. Management of society's waste has similar characteristics to other public services such as power generation, sewage treatment and major trunk roads. It is an activity from which all members of society benefit, but which also may have actual or perceived adverse environmental effects, particularly on populations living and working close to specific facilities.

This chapter sets out contextual information which may be helpful in appreciating the scale of the health and environmental issues associated with management of MSW. Annex 6.1 provides a brief discussion of public perceptions of waste management.

Communicating about health and environmental risks is a challenging problem. One approach is to use risk comparisons (see for example Department of Health, 1999). Risk comparisons are identified by Petts and Eduljee (1994) as a means by which experts have tried to improve the presentation of risk assessments to lay audiences. However, Petts and Eduljee note that when using risk comparisons the limitations of such an approach (e.g. data uncertainties, variable data quality, simple nature of assessments) should be openly addressed and the comparisons chosen should be specific and relevant to the audience.

For this reason, the information presented in this chapter could be unhelpful rather than beneficial if it is not used appropriately to the local circumstances. In using comparisons it is important to be clear about why certain comparisons are being made (otherwise suspicion will be aroused that favourable comparators have been selected to support a particular agenda) and also to emphasise what is included in a comparison and what is not (e.g. because of a lack of data). Information on the limitations of the data relating to waste management and other activities is provided throughout this report.

To assist in communicating the health and environmental risks of waste management to a wide audience, we have provided comparisons and context on a range of scales. These should be used with care to inform and focus the debate on the important aspects of waste management.

- "per tonne of waste" scale this enables the effects of a particular waste management facility of a certain size to be estimated. They can then be considered in context with the effects of a facility such as a motorway, a sewage works, a power station, or a factory
- "household" scale this enables the health/environmental effects of waste from a household to be considered in context with the effects of a car, cooking/heating, or household sewage.
- "national" scale this enables the effects of dealing with our household waste to be considered in context with activities like farming, transportation, sewage treatment, power generation, domestic emissions, emissions from different industrial sectors

Guidance on communication of risks and scientific information is given in:

• DETR/Environment Agency/IEH guidance document, "Framework for Environmental Risk Assessment", 2000 (available from <u>www.defra.gov.uk</u>)



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction	uction Co		parison between MSW manage	ment options	Cor	mparison between MSW management ar	d other activities

- Department of Health, 2000, "Communicating about risks to public health: pointers to good practice" (available from www.doh.gov.uk)
- Parliamentary Office of Science and Technology, 2000: "Science and the public: a review of science communication and public attitudes to science in Britain"
- The Integrate project (available from <u>www.etsu.com/integrate</u>)

Introduction	Review of Information on Emissions	Review of Epidemiological Research	eview of pidemiological esearch Quantification of Health Consequences of Emissions Emis		•	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	arison between MSW manage	ment options	Сог	nparison between MSW management an	d other activities

6.2 Comparison between MSW management options

This section provides comparisons between the different waste management options considered.

Emissions from MSW facilities

This chapter provides contextual information for a comparison of emissions to air, land and water. This information can be provided most fully for emissions to air. Contextual information for emissions to land and water is also provided.

The information set out in this section is relevant to the operation of waste management facilities under normal conditions. From time to time, however, any industrial facility will experience incidents when operation does not take place normally. Under these circumstances, emissions could exceed the levels set out in this section for short periods. This would not normally have a significant influence on emissions averaged over the long term. In Chapter 2, the kind of abnormal operating conditions which could occur for different processes are set out, together with an indication of how these might affect emissions. This discussion is brought together in Section 7.2.1. The evidence for the presence or absence of health and environmental effects of MSW management provided in Chapter 3 and Chapter 5 is based on facilities operating in the real world, and therefore includes any effects of operation under abnormal operating conditions.

The key atmospheric emissions identified and summarised in Chapter 2 from landfill, incineration and composting are reproduced below in Table 6.1. The existence of a larger body of information on emissions to air compared to emissions to other media should not be taken to imply that releases to other media are less significant. Instead, it may highlight a need for further research into emissions other than to atmosphere.

No information was available on emissions from materials recycling facilities (MRFs). MRFs provide an opportunity for materials in the waste stream to be recycled. Reprocessing materials in this way could result in increases or decreases in emissions at locations remote from the MRF itself. These complex issues lie outside the scope of this report, which focuses on the potential emissions and effects associated with the facility itself.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	arison between MSW manage	ement options	Co	mparison between MSW management ar	d other activities

Table 6.1 Emissions to air from waste management (grams per tonne of waste)

Substance	Windrow composting	МВТ	Anaerobic Digestion	Pyrolysis/ gasification	Mass burn Incineration	Small scale Incineration / pre-sorting	Landfill / engines	Landfill/flaring	Transportation
Nitrogen Oxides	Not likely to be emitted	72.3 M(5)	188 M(8)	780 M(8)	1600 G(9)	1587 M(7)	680 M(6)	75 M(6)	31 M(7)
Total Particulates	175 P(3)	No data	No data	12 M(8)	38 G(9)	8 M(7)	5.3 M(6)	6.1 M(6)	1.3 M(7)
Sulphur Dioxide	Not likely to be emitted	28 M(5)	3.0 M(8)	52 M(8)	42 G(9)	20 M(7)	53 M(6)	90 M(6)	0.11 M(7)
Hydrogen Chloride	No data	1.2 M(5)	<0.02 M(8)	32 M(8)	58 G(9)	74 M(7)	3 M(6)	14 M(6)	Not likely to be emitted
Hydrogen Fluoride	Not likely to be emitted	0.4 M(5)	<0.007 M(8)	0.34 M(8)	1 G(9)	1 M(7)	3 M(6)	2.7 M(6)	Not likely to be emitted
Volatile Organic Compounds	No data	36 M(5)	No data	11 M(8)	8 M(8)	33 M(7)	6.4 M(6)	7.6 M(6)	5.1 M(7)
1,1 - Dichloroethane	No data	No data	No data	Not likely to be en	nitted		0.66 M(6)	0.66 M(6)	Not likely to be emitted
Chloroethane	No data	No data	No data	Not likely to be en	nitted		0.26 M(6)	0.26 M(6)	Not likely to be emitted
Chlorothene	No data	No data	No data	Not likely to be en	nitted		0.28 M(6)	0.28 M(6)	Not likely to be emitted
Chlorobenzene	No data	No data	No data	Not likely to be en	nitted		0.59 M(6)	0.59 M(6)	Not likely to be emitted
Tetrachloroethene	No data	No data	0.0004 M(7)	Not likely to be en	nitted		0.98 M(6)	0.84 M(6)	Not likely to be emitted
Benzene	No data	No data	No data	Not likely to be en	nitted		0.00006 M(6)	0.00006 M(6)	0.0029 M(7)
Methane	No data	411 M(5)	No data	No data	19	No data	20,000 M(6)	19,000 M(6)	Not likely to be emitted
Cadmium	Not likely to be emitted	No data	<0.0001 M(8)	0.0069 M(8)	0.005 G(9)	0.007 M(7)	0.071 M(6)	0.071 M(6)	No data
Nickel	Not likely to be emitted	No data	<0.0003 M(8)	0.04 M(8)	0.05 M(8)	0.33 M(7)	0.0095 M(6)	0.0095 M(6)	No data
Arsenic	Not likely to be emitted	No data	<0.0005 M(8)	0.06 M(8)	0.005 M(8)	0.033 M(7)	0.0012 M(6)	0.0012 M(6)	No data
Mercury	No data	No data	<0.0006 M(8)	0.069 M(8)	0.05 M(8)	0.021 M(7)	0.0012 M(6)	0.0012 M(6)	No data
Dioxins and Furans	No data	4.0×10^{-8} M(5)	No data	$4.8 \times 10^{-8} M(8)$	$4.0 \times 10^{-7} G(9)$	2.4×10^{-6} M(7)	1.4×10^{-7} M(6)	$5.5 \times 10^{-8} M(6)$	$3.8 \times 10^{-11} M(7)$
Polychlorinated Biphenyls	No data	No data	No data	No data	0.0001 M(8)	No data	No data	No data	No data
Carbon Dioxide	No data	181000 M(5)	No data	No data	1000000 G(9)	No data	300000 M(6)	200000 M(6)	1170

Note: The uncertainty associated with these estimates is set out in Chapter 2. Data Pedigree:P(1-4): Poor; M(5-8): Moderate; G(9-12): Good; VG(13-16): Very Good



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	arison between MSW manage	ment options	Cor	nparison between MSW management an	d other activities

Emissions to air per tonne of key substances are shown in Figures 6.1 to 6.5:

- Figure 6.1 : Oxides of nitrogen
- Figure 6.2 : Particulates
- Figure 6.3 : Sulphur dioxide
- Figure 6.4 : Arsenic
- Figure 6.5 : Dioxins and furans
- Figure 6.1 Comparison of emissions of oxides of nitrogen to air per tonne of waste processed



Introduction	Review of Rev Information on Epic Emissions Res		view of Quantification of Health oidemiological Consequences of seearch Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	arison between MSW manage	ment options	Comparison between MSW management a	nd other activities

Figure 6.2 Comparison of emissions of particulate matter to air per tonne of waste processed



"Particulate matter" refers to both fine dusts emitted from the movement or disturbance of waste, as well as particulate matter emitted from waste combustion. The wide range in estimates of particulate emissions from incineration reflects the differences in low release concentrations in emissions from waste incinerators.



Figure 6.3 Comparison of emissions of sulphur dioxide to air per tonne of waste processed

Introduction	Review of Information on Emissions	Review of Epidemiological Research	f Quantification of Health Consequences of Emissions Quantification of the Environmental Consequences of Emissions		9	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	arison between MSW manage	ment options	Cor	nparison between MSW management an	d other activities

Figure 6.4 Comparison of emissions of arsenic to air per tonne of waste processed



Figure 6.4 shows a relatively high value for emissions of arsenic from pyrolysis/gasification. This value is based on measurements from a single facility, and the high value might just be due to an abnormally high measurement.

Figure 6.5 Comparison of emissions of dioxins and furans to air per tonne of waste processed



A relatively high rate of emissions of dioxins and furans from an incinerator with pre-sorting was derived. This information was derived from a single process - the only operational UK plant of this type – and so is of less good quality than data for mass-burn incineration.

Emissions from waste management processes involving energy recovery will be offset to a varying extent by emissions avoided from the use of fossil fuels. While a



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Review of Quantification of Health Epidemiological Consequences of Emissions Emissions Quantification of the Environmental Consequences of Emissions			Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	arison between MSW manage	ment options	Cor	nparison between MSW management an	d other activities

detailed evaluation of this lies outside the scope of this project, we have carried out an initial assessment of offset emissions. This suggests that anaerobic digestion performs best of the facilities which have an energy generation component, resulting in net reductions in emissions of most pollutants when avoided emissions from energy generation are taken into account. Energy generation from waste management generally gives rise to reductions in overall emissions of sulphur dioxide and particulates, when emissions from power stations are taken into account. Emissions of other pollutants from waste management are likely to give rise to no significant change or a net increase in emissions.

Information on emissions to surface water and groundwater from waste management activities was gathered as part of this project. Emissions are currently dominated by landfill, because other processes generate little or no liquid effluent. Information on releases to surface water and groundwater from landfill in context with total UK emissions is therefore provided in the following section.

Health effects of MSW facilities

The health effects of MSW management facilities are set out in Chapter 4. Figures 4.1 to 4.5 show the health effects of emissions to air per tonne of waste processed. Figure 6.6 shows a comparison of the estimated numbers of deaths brought forward as a result of emissions from an individual facility for the facilities studied.





Introduction	Review of Information on Emissions	Review of Epidemiological Research	eview of Quantification of Health pidemiological Consequences of esearch Emissions			Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Cor	parison between MSW manage	ement options	Com	nparison between MSW management an	d other activities

6.3 Comparison between MSW management and other activities

This section sets out a comparison between the emissions, health and environmental effects of waste management, and those associated with other activities.

Emissions from MSW management in context

The majority of MSW in the UK is disposed via landfill and incineration. UK emissions to air from these two routes for 2000/2001 are set out in Table 6.2.

Substance	Emission (tonnes per	year based on from	2000/2001)	Pedigree
	Incineration	Landfill	Total	
Nitrogen Oxides	3892	6,091	9,983	Moderate (6)
Total Particulates	92	104	196	Moderate (6)
Sulphur Dioxide	102	1,896	1,998	Moderate (6)
Hydrogen Chloride	141	259	400	Moderate (6)
Hydrogen Fluoride	2.4	67	69	Moderate (6)
Volatile Organic Compounds	19	251	270	Moderate (6)
Non-methane VOCs	No significant emission	201	201	Moderate (6)
1,1 - Dichloroethane	No significant emission	24	24	Moderate (6)
Chloroethane	No significant emission	9	9	Moderate (6)
Chlorothene	No significant emission	10	10	Moderate (6)
Chlorobenzene	No significant emission	21	21	Moderate (6)
Tetrachloroethene	No significant emission	31	31	Moderate (6)
Benzene	No significant emission	2	2.0	Moderate (6)
Methane	46	694,000	694,000	Moderate (6)
Cadmium	0.012	0.51	0.52	Moderate (6)
Nickel	0.12	0.068	0.189636	Moderate (6)
Arsenic	0.012	0.0086	0.020764	Moderate (6)
Mercury	0.12	0.0086	0.130236	Moderate (6)
Dioxins and Furans	9.7 × 10 ⁻⁷	$1.9 imes 10^{-6}$	$2.9 imes10^{-6}$	Moderate (6)
Polychlorinated Biphenyls	0.00024	-	0.000243	Moderate (6)
Carbon Dioxide	2,430,000	1,180,000	3,610,000	Moderate (6)

Table 6.2 Comparison of emissions to air from waste management activities (2000/2001)

1. The total UK MSW arisings for 2000/2001 used in the calculations was taken from the Digest of Environmental Statistics for England and Wales produced by DEFRA <u>http://www.defra.gov.uk/environment/statistics/des/index.htm;</u> the Waste Digest for Scotland produced by SEPA <u>http://www.sepa.org.uk/publications/wds/index.htm;</u> and the Municipal Waste Arisings Survey for Northern Ireland produced by the Environment and Heritage Service <u>http://www.ehsni.gov.uk/publcations/MunicWasteSurvey2001_ExecSumm.pdf</u>.



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Com	parison between MSW manage	ment options	Cor	nparison between MSW management ar	d other activities

Table 6.3 summarises the UK emissions of air pollutants for 2000 from the National Atmospheric Emissions Inventory (NAEI), compared to waste management operations and other activities.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	9	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	parison between MSW manage	ment options	Co	mparison between MSW management an	d other activities

Substance	Units	Total emissions	Power production	Road transport	Management of MSW	Agriculture	Domestic	Other
Carbon dioxide as C	Te y ⁻¹	147,500,000	42,000,000	31,500,000	3,600,000	200,000	23,400,000	
Methane	Te y ⁻¹	2,427,000	28,000	16,000	690,000	969,000	29,000	
PM10	Te y ⁻¹	172,000	22,000	26,000	200	14,000	28,000	
Benzene	Te y ⁻¹	16,430		7710	2.0		3000	
Oxides of nitrogen	Te y ⁻¹	1,512,000	358,000	629,000	10,000		72,000	
Sulphur dioxide	Te y ⁻¹	1,165,000	826,000	6000	2,000		44,000	
Hydrogen chloride	Te y ⁻¹	84,000	72,400		400		4500	
NMVOC	Te y ⁻¹	1,676,000	8000	408,000	201		36,000	
Hydrogen fluoride	Te y ⁻¹	3800	2000		69		200	
Dioxins and furans	G TEQ y ⁻¹	360	14	12	2.9		65	50 (fireworks, estimated: <u>www.nsca.org.uk</u>)
Polychlorinated biphenyls	kg y⁻¹	1706	49		0.24		10	1200 (fluid in old electrical equipment)
Arsenic	Te y ⁻¹	34.6	4.3		0.021		7.3	
Cadmium	Te y ⁻¹	5.2	0.4	0.4	0.52		0.3	
Lead	Te y ⁻¹	496	17.5	326			13.6	
Mercury	Te y ⁻¹	8.5	1.4		0.13		0.5	1.3 (Crematoria)
Pedigree		Metals, VOCs, dioxins, PCBs Poor Others: Moderate	NO _x , SO ₂ , CO ₂ Good Others: Moderate	Metals, dioxins: Moderate Others: Good	Moderate	Poor	Poor	

Table 6.3 Comparison of emissions to air from waste management and other common activities (National Atmospheric Emissions Inventory, 2000)



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	•	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction			arison between MSW manage	ment options	Cor	mparison between MSW management ar	d other activities

The following conclusions can be drawn from Table 6.3:

- The electricity supply industry is the major source of carbon dioxide emissions in the UK as it is the major consumer of fossil fuels. Total emissions from transport equate to around 24% of the total UK carbon dioxide emissions. Good quality information is available on these emissions
- The largest source of methane emissions is the agricultural sector. Emissions from landfill are estimated to account for approximately 27% of UK methane emissions in 2000, although this estimate is of poor quality. The value for waste management derived in this project is dependent on assumptions regarding the different types of landfill site in the UK and the variation in methane emissions during the lifetime of a landfill. However, the estimated emission of 690,000 tonnes per year is consistent with the value of approximately 550,000 tonnes per year identified in a more detailed study focusing specifically on methane emissions from landfill (LQM, 2002). These values are of moderate quality. Methane emissions are likely to decrease in the future as extraction and combustion of landfill gas increases, and the biodegradable content of landfilled material decreases.
- Emissions of benzene are dominated by transport, accounting for 47% of the 2000 estimate total. The estimate of emissions from traffic is of good quality, but estimated emissions from other sources are only considered to be of poor or moderate quality.
- PCBs have not been manufactured and used in the UK for many years, but it is estimated that 81% of PCB emissions are associated with PCB-containing equipment that still exist. Large quantities of PCBs are thought to have been disposed of to landfill in the past.
- The largest emission of arsenic arises from coal combustion with other sources being very small by comparison. Coal use has declined over the period considered, in favour of natural gas use. The emissions from the industrial sector are large compared with the emissions from public power generation; this is due to the different levels of abatement efficiency that are assumed. The estimated emissions of arsenic are of moderate quality.
- The main sources of cadmium are non-ferrous metal production and iron and steel manufacture.
- The largest source of lead is road transport, although the introduction of leadfree fuels has reduced emissions. Other major sources are industrial processes and iron and steel manufacture.
- The main emissions of mercury are from waste incineration, cremation, the manufacture of chlorine in mercury cells, non-ferrous metal production and coal combustion.
- Estimated emissions of dioxins and furans from management of MSW account for about 1% of the UK total, shared approximately equally between incineration and landfill gas combustion. A number of sources contribute to emissions of dioxins and furans to a similar or greater extent: accidental vehicle fires; smallscale waste burning (e.g. on building sites); incineration of other wastes; and the iron and steel industry. However, the most significant sources of dioxins and furans are domestic emissions and fireworks, both of which account for about a sixth of total emissions. Information on emissions of dioxins from waste



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction Co			Comparison between MSW management options		Com	nparison between MSW management an	d other activities

management and power generation is of moderate quality; information on emissions from other sources is of poor quality.

A different perspective can be gained by considering how much road traffic would give the same emissions as managing MSW. The national UK emissions of oxides of nitrogen from management of MSW are approximately equivalent to emissions of oxides of nitrogen from traffic using a motorway 200 km in length. Similarly, emissions of particulates from management of MSW are approximately equivalent to emissions from a motorway 120 km long, and emissions of carbon dioxide from management of MSW are approximately equivalent to emissions from a motorway 120 km long, and emissions of carbon dioxide from management of MSW are approximately equivalent to emissions from a motorway 500 km long.

Figure 6.6 presents a graphical comparison of the data in Table 6.3. The reader is asked to pay particular attention to the units associated with each substance, as these differ from one substance to another. In the majority of cases emissions from disposal of MSW are a small fraction of the UK total. Exceptions to this are waste management emissions of methane and cadmium which account for 27% and 10% respectively of the total UK emissions. The majority of estimated cadmium emissions to air from waste management arise from landfill.



Figure 6.7 Comparison of emissions to air from waste management and other activities

Table 6.4 presents a breakdown of the emissions attributable to UK households, broken down to an average household level. Approximately 1300 kg of household waste was generated per household in 2000/2001, equivalent to 26 kg per household per week (DEFRA, 2001).



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	5	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	Comparison between MSW management options		Cor	nparison between MSW management ar	d other activities

Table 6.4 Breakdown of emissions to air per household (kg y⁻¹)

	Cooking 8	Domostic	Domestic machinery	Carl	MSW	/ manageme	ent
Substance	heating	products ²		motorcycle	Total	Landfill	Inciner -ation
Carbon dioxide as C	950 (M)			212 (G)	147 (M)	48 (M)	99 (G)
Methane	1 (P)			0.50 (M)	28 (M)	28 (M)	-
NO _x	3 (M)		0.04 (M)	15 (G)	0.41 (M)	0.25 (M)	0.16 (G)
NMVOC	1 (M)	3 (P)	1 (P)		0.0089 (M)	0.0082 (M)	0.0007 (M)
Sulphur dioxide	1 (M)		0.001 (M)	0.19 (G)	0.081 (M)	0.077 (M)	0.004 (G)
PM ₁₀	1 (P)		0.01 (P)	0.49 (M)	0.008 (M)	0.004 (M)	0.004 (G)
Benzene			0.02 (P)	0.29 (M)	0.000082 (M)	0.000082 (M)	-

- 1. Data pedigree: VG: Very good; G: Good; M: Moderate; P: Poor
- There are 21,660,475 households in England and Wales and 2,203,000 in Scotland according to Census 2001. There are approximately 700,000 households in Northern Ireland (derived from Waste Management Strategy Northern Ireland)
- Cooking and heating includes the consumption of burning oil, gas oil, fuel oil, coke, coal, anthracite, solid smokeless fuels, LPG, natural gas. LPG assumed to be consumed a 1% rate of natural gas, wood assumed to be consumed at the same rate as coal. Emission rates for cooking and heating, domestic products (e.g. cleaning products, cosmetics and paint thinner) and domestic machinery (e.g. garden equipment such as petrol engine lawnmowers) were obtained from the National Atmospheric Emissions Inventory and fuel consumption rates from UK Digest of Energy Statistics <u>http://www.aeat.co.uk/netcen/airqual/naei/annreport/annrep99/app1_22.html</u> http://www.dti.gov.uk/epa/digest01/contents01.htm.
- 4. Car and motorcycle emissions are for all UK car and motorcycle emissions (i.e. this includes a proportion of business related emissions)

From the data presented in Table 6.4 it can be seen that waste management makes a significant contribution to household emissions of greenhouse gases (carbon dioxide and methane; data of moderate quality). Use of cars and motorcycles makes the greatest contribution to emissions of oxides of nitrogen and benzene (data of good quality). The use of products such as aerosol and non-aerosol cleaners and cosmetics is estimated to result in much greater emissions of nonmethane volatile organic compounds per household than disposal of waste. However, the estimated emissions from cleaning products and cosmetics are of poor quality.

Table 6.5 compares emissions to surface water and groundwater from landfill sites using data presented in Chapter 2 and the total UK emissions. Emissions from landfill sites are estimated to account for between 0.005% and 0.32% of total UK sources, although these estimates are of poor quality.

DEPARTMENT FOR ENVIRONMENT, FOOD AND RURAL AFFAIRS

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	•	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction Corr			parison between MSW manage	ment options	Co	nparison between MSW management ar	d other activities

Table 0.5 Comparison of emissions (re y) water from fanding with other source	Table 6.5	Comparison of emis	sions (Te y ⁻¹)	water from	landfill with	other sources
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Substance	Landfill releases	Landfill re	leases to sewer	All releases to	
	to groundwater/ surface waters	To STW	From STW to receiving waters	UK coastal waters	
Total nitrogen	232	1062		400,000	
Organo-tin compounds	0.00013	0.00058	0.0000023	0.45	
Phosphates	1.9	8.7		33000	
Pentachlorophenol	<0.000026	<0.00025	<0.000035	3.4	
Copper	0.0035	0.027	0.0091	590	
Lead	<0.032	<0.15	<0.071	480	
Zinc	0.026	0.31	0.12	2400	
Pedigree	Poor (4)	Poor (4)	Poor (3)	Poor	

1. Data for releases to UK Coastal Waters are averages of estimates of releases taken from the Digest of Environmental Statistics for England and Wales produced by DEFRA <u>http://www.defra.gov.uk/environment/statistics/des/index.htm</u>, and include discharges via river water and substances naturally occurring in river water

Health effects of MSW management in context

The key potential health impacts identified and summarised in chapter 4 are shown below in Table 6.6. Table 6.6 also shows the health impacts associated with some other common causes of disease. A description of the numbers of people likely to be affected by these causes based on a community risk scale is also given in Table 6.6. Table 6.7 sets out some other common risks to health.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	•	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction		Comp	Comparison between MSW management options		Co	mparison between MSW management ar	d other activities

Table 6.6 Comparison of health impacts from waste management and other causes of disease

Health		Numb	per per year in the U	IK ¹ due to	
impact	Landfill emissions to air (estimated)	Incineration emissions to air (estimated)	Skin cancer (malignant melanoma: main UK causes are sunlight and sunbeds) ²	Lung cancer due to passive smoking ³	Health impacts due to air pollution ⁴
Deaths brought forward	0.4 (less than one nationally per year)	0.15 (less than one nationally per year)			11,600 (about one per small town per year)
Hospital admissions	0.9 (Respiratory) 0.03 (Cardiovascular) About one nationally per year	4 (Respiratory) 0.0010 (Cardiovascular) About one per region per year			14,000 (about one per small town per year)
Cancers	0.0013 (about one nationally every seven hundred years)	0.0005 (about one nationally every two thousand years)	6,000 (about one per small town per year)	"several hundred" (about one per large town per year)	
Data Pedigree	Poor quality	Moderate quality	Moderate quality	Poor quality	Poor quality

- 1. Community risk scale taken from Department of Health, 2000, "Communicating about risks to public health: pointers to good practice"
- 2. Information taken from National Radiological Protection Board, 2003
- 3. Information taken from Committee on Carcinogenicity, 1998
- 4. Information taken from COMEAP, 1998 (acute effects of PM_{10} and sulphur dioxide in urban areas)

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	9	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction Co			parison between MSW manage	ment options	Co	mparison between MSW management ar	d other activities

Table 6.7	Health	impacts	from	other	common	activities
	meantin	impacts		other	common	activities

Health impact			Number per y	/ear due to		
	Accidents in the home	Accidents in the workplace	Road traffic accidents (2000)	Natural or environmental factors (e.g. excessive cold) (2000)	Choking on food (2000)	Injury from fireworks (2002)
Deaths	4300	736	3,409	191	246	
brought forward	One accidental death per small town	One accidental death per large town	One accidental death per small town	One death per large town	One death per large town	
Hospital admissions	168,300 One hospital admission per street or village	500,000 (approximately) One hospital admission per street	320,000 One hospital admission per street			1017 One hospital admission per small town
Quality	Good	Good	Good	Good	Good	Good

- 1. Data on accidents from the Royal Society on Prevention of Accidents "General Accidents" factsheet, available from http://www.rospa.com
- 2. Community risk scale taken from Department of Health, 2000, "Communicating about risks to public health: pointers to good practice"

Environmental effects of MSW management in context

The literature review (Chapter 5) identified a limited range of quantitative information on the environmental impacts of waste management which can be set in the context of other activities affecting the environment. The available information is less detailed than that provided by the review of emissions data or information on health effects.

The majority of comparisons identified from the literature relate to incineration:

- In the vicinity of a source in an area remote from other sources, the source contribution may be greater than the national contribution of the source type as a proportion of national emissions. Even so, other sources of dioxins in soils and herbage close to an MSW incinerator have been found to be more significant than emissions from the incinerator. One study reported 0.54 ng/kg in soils due to incinerator emissions, compared to a typical background level of 10 ng/kg.
- Similarly, forecast levels of dioxins due to incinerator emissions in the immediate vicinity of an incinerator have been reported as between 10% and 100% of prevailing background levels. In terms of overall public exposure, incinerators are reported as accounting for 0.8% of uptake of dioxins and furans (this estimate is of moderate quality)



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions	
Introduction		Com	Comparison between MSW management options		Comparison between MSW management and other activities		

- A statistical analysis of the variation in levels of metals in snow and rainwater indicated that less than 20% of the variation could be accounted for by emissions from a waste incinerator. This suggests that other sources/influences are more important in determining levels of metals in precipitation.
- 4% of mercury in US originates from waste incinerators (however, more up to date UK information is provided above).
- Mercury emissions from a MSW incinerator were found to account for about one fifth of background ground level concentrations.
- Road traffic was found to have a more significant effect on local levels of PAHs than an incinerator

Noise levels

Quantitative data are provided on recorded incremental noise levels close to waste management facilities. This information is of good quality.

- Composting +7 to +32 dBA increment recorded.
- Materials recycling facility +15 to +20 dBA increment recorded.
- Landfill +5 to +10 dBA increment recorded
- Gasification/ pyrolysis +5 dBA increment recorded.

Typical incremental noise levels associated with other sources are as follows (data of good quality; derived from Environment Agency, 2003):

- Noise level associated with conversation at 1 metre
 c.+20 dBA increment
- Noise level associated with office activity: c.+25 dBA increment
- Noise level associated with vacuum cleaner at 3 metres c.+35 dBA increment

Noise and odour complaints

From the data provided in Chapter 6, the numbers of complaints associated with odour and noise from landfills can be estimated (estimates are of moderate quality):

- Odour: estimates of between 4,000 and 10,000 complaints per year nationally
- Noise: approximately 300 complaints per year nationally

Information from the Chartered Institute of Environmental Health (CIEH, 2003) indicates that a total of between 12,000 and 30,000 complaints of odour are received annually by local authorities. An accurate total cannot be estimated because of incomplete reporting to the CIEH, and the unpredictable pattern of complaints (e.g. a single incident may generate one complaint, or hundreds of complaints). This estimate is therefore of poor quality.

The total number of noise complaints recorded by the CIEH is approximately 400,000 per year. This value is of moderate quality.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	9	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions	
Introduction		Comp	Comparison between MSW management options			Comparison between MSW management and other activities		

This indicates that landfills account for a significant proportion of odour complaints, likely to be between 10% and 25% of all odour complaints received by local authorities (this value is of poor quality). Landfills and other waste management operations are unlikely to account for a significant proportion of noise complaints.
Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Introduction	1	Comp	omparison between MSW management options		Cor	nparison between MSW management an	d other activities

Annex 6.1 : Public perception of waste management

A survey carried out in 2002 (Eurobarometer 58.0, European Commission, 2002) suggested that waste management was relatively low down the list of public environmental concerns. In the UK, 24% of those surveyed were "very worried" about domestic waste, and 37% "very worried" about industrial waste. Domestic waste management is the 18th highest concern, and industrial waste management the 11th highest concern out of 25 listed.

In view of this, it is perhaps surprising that public attitudes to specific waste management facilities are so strongly negative. The reasons why waste management facilities are viewed in this way include the following:

- Waste management facilities have many of the characteristics listed above as increasing the unacceptability of the associated risks –
 - The risks are involuntarily imposed
 - The risks are unfamiliar (e.g. chemicals or micro-organisms with obscure names in particular dioxins)
 - Some of the risks pose a threat of a dreaded form of death or illness such as cancer or birth defects
 - The risks arise from man-made facilities and materials in the waste
 - The risks may not be apparent to local residents, and may be delayed
 - The risks are focused around the individual facilities, whereas the benefit of the facilities is shared across society as a whole
 - Some of the risks affect children and future generations
 - The health and environmental risks of waste management have been the subject of controversy and contradictory information e.g. from the waste industry and environmental pressure groups.
- Particular attention has been focused on emissions of dioxins from waste management facilities, sometimes to the extent that a balanced message is not communicated. For example, other emissions from waste management facilities which may also be of concern may not be considered in as much detail. Also, improvements in assessment and control of emissions of dioxins from waste management facilities, and the significance of other sources of dioxins are not always appreciated.
- We become instinctively aware that waste is dirty and unhealthy, partly as a survival mechanism to avoid disease from direct contact with waste. This attitude is also applied to waste management facilities.
- There is a lack of appreciation by the public of what is involved in waste management. A study by the National Society for Clean Air and Environmental Pollution (NSCA, 2001) noted that "public understanding and awareness of waste issues is currently very poor". A similar conclusions was noted in a survey carried out for the Environment Agency Hazardous Waste forum.



Introduction	Review of Information on Emissions	Review of Epidemiolo Research	ical	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions	9	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction	1		Comp	parison between MSW manage	ment options	Comparison between MSW management and		d other activities

- Waste management facilities can in some cases give rise to odours which are very noticeable to local communities. It is natural to associate odours with the risk of health problems.
- In common with all industries, there have been environmental problems at some waste management facilities. This can increase the profile of the problems, so that the whole industry becomes "tarred with the same brush" although the problems may not be relevant to the majority of well run facilities.
- A further problem common to any new development is the desire for individuals to oppose new industrial-scale facilities close to their own home.
- There is a general increase in distrust of scientists and scientific advice, particularly in the aftermath of the BSE and foot and mouth epidemics. Nevertheless, the Eurobarometer survey indicated that 33% of those surveyed trusted scientists for environmental information. 38% trusted environmental associations, whereas only 7% trusted national government. The Parliamentary Office of Science and Technology (POST, 2000) carried out a study of public attitudes to science which highlighted concerns over the ability of society to control science. People were concerned about science being carried out behind closed doors. 43% of those surveyed believed that the benefits of science are greater than any harmful effects; 17% did not believe this; and one third were undecided.
- With the economic shift away from manufacturing industry towards the service economy, there are relatively few new industrial facilities being constructed at present. Waste management is one of the few areas where new facilities are being proposed, and this results in their being the focus of greater attention.
- The presence of a waste management facility is perceived as resulting in a decrease in the price or saleability of a property. It is likely that in some cases, environmental and health concerns are given as the reasons for opposing a particular development, when there is also an unspoken concern with regard to property values.

The Environmental Technology Support Unit "Integrate" project compared UK experience of public attitudes to energy from waste schemes with experience elsewhere in Europe. It was found that the more positive attitude to waste management schemes found in countries such as Denmark, the Netherlands and Sweden was helped by a number of factors:

- Familiarity with issues relating to energy, waste, and the proposed technology
- Industrialised areas were more open to the development of energy from waste schemes
- If a new scheme can be introduced without an increase in heavy goods vehicles (e.g. by using water-borne transport, or replacing an existing traffic-generating land use), this can deal with one of the major issues of public concern
- Meeting local needs for waste disposal and energy generation
- Trust between the developer and local communities
- Involvement or commitment of the local council
- Confidence in the regulator

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quantification of the Environmental Consequences of Emissions		Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	Conclusions
Introduction	1	Comp	arison between MSW manage	ment options	t options Comparison between MS		d other activities

- Integrated and flexible waste strategy
- Fitting in with a local vision

Public concern relating to specific proposals is frequently triggered in two ways

- The agenda set by national and local pressure groups. On a national scale, this tends to be focused on encouraging waste minimisation and recycling, and on the health and environmental effects of waste incineration. Less attention is paid to landfill, and little or no attention is given to the health and environmental effects of other waste management technologies.
- Local issues can also trigger public concern, such as odours, noise, dust, flies, vermin or traffic-related problems perceived to be associated with existing facilities

Guidance on communication of risks and scientific information is given in:

- DETR/Environment Agency/IEH guidance document, "*Framework for Environmental Risk Assessment*", 2000 (available from <u>www.defra.gov.uk</u>)
- Department of Health, 2000, "Communicating about risks to public health: pointers to good practice" (available from www.doh.gov.uk)
- Parliamentary Office of Science and Technology, 2000: "Science and the public: a review of science communication and public attitudes to science in Britain"
- The Integrate project (available from <u>www.etsu.com/integrate</u>)

The Integrate project report highlights the principles for good practice in communication (taken from Petts et al., 1996):

Dialogue, not monologue

Different values, interests and concerns are legitimate

Participants should seek to understand the values in which each other's interest and concerns are grounded

Public debate makes a positive contribution to decisions

Those who hold information must offer it to other stakeholders without waiting to be asked

Communicators should plan communication activity so that adequate time is allocated

Stakeholders should ensure people are trained in communication

Stakeholders should regularly evaluate their communication activities: i.e., have we met our objectives? Have other stakeholders met theirs? What should we be doing differently?

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Envi Con: Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	ed Health and s, & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendati future work	ons for

7. CONCLUSIONS

248

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quar Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	Public Perception Issues cts Recommendation future work		ons for

Summary

Most emissions to air can be quantified with at least a moderate level of confidence, although there are some exceptions. The main gaps are emissions from composting, mechanical biological treatment and anaerobic digestion. We found that information on emissions to sewer, surface water, groundwater and land is more patchy. However, we were still able to gather useful information on these emissions.

The information we have gathered enables emissions from different processes to be compared. It also enables us to compare emissions from waste management with emissions from other widespread sources of pollution. Emissions from landfill differ from other waste management processes because they occur over a period of years.

We looked at evidence for ill-health in people who might possibly be affected by emissions from MSW processes. For most of the municipal solid waste facilities studied, we found that health effects in people living near waste management facilities were either generally not apparent, or the evidence was not consistent or convincing. However, a few aspects of waste management have been linked to health effects in local people. We would need more research to know whether or not these are real effects. We also investigated the health effects of emissions of some important airborne pollutants from waste management facilities. Although the data was of moderate or poor quality, we found that these emissions are not likely to give rise to significant increases in adverse health effects.

We have recommended areas where further research would be helpful to fill the gaps in what we know at present.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quar Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Iss	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues	
Introduction	Emissions		Health Effects		Environmental Effe	Environmental Risks, & Review of Public Perception Issues ts Recommendation future work		ons for

7.1 Introduction

We have carried out a detailed review of published information on the health and environmental effects of the transport, treatment and disposal of municipal solid waste (MSW) and similar wastes.

The main component of the study was a review of scientific literature and other published information on emissions from MSW management, and the health and environmental effects of MSW management. This drew on papers published in the peer-reviewed literature; research carried out by governmental and non-governmental organisations, and information from the operators of waste management processes (often provided to the regulator as part of their operating licence requirements).

This review enables us to estimate many of the emissions from waste management operations, as a quantity of each substance emitted per tonne of waste processed. Using this information, we were able to estimate the quantities emitted by an individual facility and to derive a national total for these emissions. We were also able to highlight areas where MSW management operations may give rise to health effects, and areas where no health effects have been found. We quantified the significance of some of these effects.

We then considered the relative performance of different kinds of waste processing and disposal operations, and the environmental and health effects of MSW management compared to other activities. Finally, we considered where further research could usefully be carried out to improve our understanding of the relationship between waste processing and adverse environmental and health effects.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Envi Con: Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation	ons for

We found that management of MSW currently contributes less than 2.5% to UK emissions to air (data quality mostly moderate; in some cases good, and in some cases poor), and less than 0.5% to UK emissions to water (data quality poor). The exceptions to this are emissions to air of methane (where MSW processing and disposal accounts for 27% of UK total emissions) and cadmium (MSW management accounts for 10% of UK total emissions).

Studies have been carried out to investigate whether the occurrence of certain birth defects (known as *congenital anomalies*) might be explained by the presence of landfill sites. There is some evidence to suggest that these defects occur slightly more often in children born to mothers living close to a landfill site. We cannot yet say whether the landfills cause or contribute to this apparent clustering of birth defects. Factors which include the mother's health and the child's genetic make-up are known to be causes of birth defects, but even so, the majority of birth defects are of unknown origin. Increases in some respiratory diseases have also been observed in people living very close to composting facilities.

We calculated the probable health effects of some key pollutants emitted to air from waste management operations. While the numerical estimates are of moderate or poor quality, we found that the probable health effects of these emissions are very small in comparison to those of other common hazards to health.

We found that relatively reliable information exists for emissions to air from MSW combustion processes such as incineration and landfill gas flares/engines. Information on emissions to water and groundwater is of less good quality, because of the assumptions which need to be made to estimate these emissions. We also need more information on emissions to air and water from composting MSW.

This chapter sets out conclusions on the health and environmental effects of waste management which are supported by the evidence in the preceding chapters. We also suggest where extent and reliability of currently available information could usefully be improved.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantified Health and Environmental Risks, & Review of Public Perception Issues		Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation future work	ons for

7.2 Emissions

7.2.1 Emissions to air

Evidence exists to enable emissions to air to be quantified for most of substances considered in this report, and for the most significant of present-day waste management operations. The existence of a considerable body of information on emissions to air compared to emissions to other media should not be taken to imply that releases to other media are less significant. Instead, it may highlight a need for further research in this area.

For some sources and substances, this can be achieved with a moderate or good level of confidence.

- Emissions which can be quantified with a good level of confidence:
 - Combustion gas emissions from incineration of MSW
- Emissions which can be quantified with a moderate level of confidence:
 - Combustion gas emissions from Mechanical Biological Treatment (MBT)
 - Combustion gas emissions from anaerobic digestion
 - Emissions from pyrolysis/gasification of MSW
 - Emissions of metals from mass burn incineration of MSW
 - Emissions from small-scale incineration of pre-sorted MSW
 - Emissions from landfill of MSW
 - Emissions from transportation of MSW
- Emissions which can be quantified with a poor level of confidence:
 - Particulate emissions from composting of MSW
- Emissions which cannot be quantified
 - Emissions from composting of MSW, other than particulates
 - Emissions of volatile organic compounds (VOCs) and metals from MBT
 - Emissions of VOCs and metals from anaerobic digestion of MSW

The information on emissions was taken mainly from UK sources, although the literature review covered international research and information sources. To the best of our knowledge, there is no information available in the UK or overseas to enable emissions from composting (other than particulate matter), MBT or anaerobic digestion to be quantified.

More information was identified for emissions to air than for emissions to other media. This is reflected in the level of detail given in this report. This does not necessarily imply that the health and environmental impacts of emissions to air are



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quar Envir Cons Emis	ntification of the ronmental sequences of ssions	fication of the nmental quences of ons Context for Quantified He Environmental Risks, & R Public Perception Issues		Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	cts Recommendatio future work		ons for

more significant than other releases. It may suggest that releases to other media should be more thoroughly investigated.

Needs for further information

In view of the information set out above, further work needs to be focused in particular on understanding emissions from composting (in particular, particulate matter/dust, bio-aerosols and VOCs). This would be helpful to investigate whether emissions from composting could make a significant contribution to the recently reported health effects in populations living close to composting facilities. Emissions from mechanical biological treatment and anaerobic digestion of MSW should also be more fully characterised. This is particularly important as it seems likely that these processes will become increasingly important for treatment of MSW in the future, and proper understanding of emissions is important to ensure that facilities can be properly sited, and control and abatement systems can be designed.

Emissions from incineration, landfill, pyrolysis/gasification and transportation of MSW are mostly understood with a moderate or good degree of confidence. Ongoing measurements will continue to be required for managing emissions from individual facilities, but further data is not likely to enable the emissions estimates to be improved much further. The uncertainty around these emissions estimates is mostly due to differences between processes, rather than to any lack of data.

Emissions to air from landfill differ from emissions from processes in the timescale over which emissions take place. Emissions from processes other than landfill occur at approximately the same time as the treatment of the waste. Emissions from landfill take place over a much longer timescale. Landfill gas generation rates are likely to be greatest during and immediately following the operational lifetime of the site (typically 20 - 30 years from starting to fill the site). During this time, landfill sites must be managed and regulated to ensure that emissions to air are controlled at acceptable levels in the local context.

There is inevitably much less information available on emissions to all media under non-standard operating conditions, compared to information on emissions under normal operating conditions. Further information on emissions under non-standard operating conditions of particular concern may be helpful. This would include for example emissions from windrow composting when waste has not been properly turned.

Comparison of emissions to air between waste management options

The following comparisons can be drawn between estimated emissions to air from different waste management options:

- Incineration generates the greatest emissions of oxides of nitrogen, followed by pyrolysis/gasification and landfill with energy generation (information of moderate or good quality).
- The greatest emissions of particulates per tonne of MSW probably arise from composting, but the data are of poor quality and cannot be wholly relied upon. Incineration is the next most significant source of particulates (information of moderate quality). Transport of waste is probably not a significant source of particulates.



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of sions Context for Quantified Environmental Risks, Public Perception Issu		ed Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	fects Recommendati		ons for

- Emissions of sulphur dioxide per tonne of waste processed are similar for all combustion-based processes (information of moderate or good quality). Transport of waste is not a significant source of sulphur dioxide.
- Emissions of hydrogen chloride and hydrogen fluoride are higher from combustion processes. Of the MSW treatment/disposal facilities studied, incineration is the most significant source of hydrogen chloride.
- Emissions of VOCs are probably more significant from landfill, composting and MBT than from combustion processes; however, we have found no data to enable us to establish the significance of VOC emissions from composting and MBT. Methane emissions, and hence global warming impacts, are greatest from the landfill options (information of moderate quality), although emissions from home composting and poorly run composting operations may also be significant.
- Emissions of metals show a mixed pattern between the different combustion processes. The available information suggests that emissions of metals to air are generally lower from anaerobic digestion than from other options (information of moderate quality).
- Reported emissions of dioxins and furans per tonne of waste from incineration of pre-sorted wastes are higher than emissions from other options. This information, however, comes from a single plant, and may reflect the abatement at this particular facility. Mass burn incineration is the next most significant source, with other combustion sources resulting in lower emissions (information of moderate or good quality).
- Emissions to air from MSW incineration in the UK have changed considerably over the past 20 years, in response to increasingly strict legislation on emissions. For example, emissions of dioxins per tonne of waste from MSW incineration have decreased by 99.8% over this period. This change was driven by increasingly stringent limits on emissions from MSW incineration plant set in European directives. To comply with the new emissions limits, MSW incineration facilities were fitted with upgraded abatement plant, or otherwise were shut down.
- Materials Recycling Facilities provide an opportunity for materials in the waste stream to be recycled. Reprocessing materials in this way could result in increases or decreases in emissions at locations remote from the MRF itself. These complex issues lie outside the scope of this report, which focuses on the potential emissions and effects associated with the facility itself.
- The likelihood and significance of emissions under non-standard operating conditions vary from one type of facility to another. We recommend that further work should be carried out to investigate these emissions. The areas of greatest concern, approximately in decreasing order of significance, are as follows:
 - Emissions from **open windrow composting** of MSW can be increased if the waste is not turned properly to allow full aeration of wastes. This is a significant concern in the light of indications of adverse health effects for people living very close to commercial scale composting facilities. It would be less of a concern for in-vessel composting of MSW because of the improved control on aeration, and opportunities to treat air extracted from invessel systems.



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	ed Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects Recommendatio		ons for

- Inappropriate use or disposal of ash residues from **incineration** of MSW has led to a risk of increased human exposure to substances in the ash. Investigations of these incidents showed that were no significant health risks associated with these particular incidents, but there is a need for detailed attention to ensure that ash residues are disposed of properly.
- Increased emissions from landfill under non-standard operating conditions are also a potential cause for concern. Leachate can leak through the lining of a landfill, but this is normally detected and controlled through borehole monitoring, with the result that it is very rare for such releases to affect surface waters. A failure in landfill gas combustion could result in increased emissions of landfill gas: emissions under these circumstances would be represented by the emissions from fugitive landfill gas set out in Table 2.34. If combustion of gas generated from anaerobic digestion of MSW were to fail, this could result in increased emissions of unburnt biogas. The biogas released would be similar in composition to landfill gas.
- Increases in emissions from **incineration or pyrolysis/gasification** of MSW under non-standard operating conditions could potentially be significant, in view of the range of substances potentially emitted. However, in practice it appears that emissions of substances of concern from the perspective of potential health or environmental effects above permitted levels occur infrequently. The lack of consistent evidence of adverse health effects associated with MSW incineration indicates that emissions under routine and non-standard conditions does not give rise to consistently detectable health effects.
- Excessive delays before treatment or disposal of waste at **any facility** could give rise to increased emissions of odour and micro-organisms. The significance of these emissions is lower at facilities equipped with air extraction and treatment systems such as dust filters or bio-filters, or where air can be passed through a combustion system (provided the combustion system is operational).

Emissions from waste management processes involving energy recovery will be offset to a varying extent by emissions avoided from the use of fossil fuels. While a detailed evaluation of this lies outside the scope of this project, we have carried out an initial assessment of offset emissions. This suggests that anaerobic digestion performs best of the facilities which have an energy generation component, resulting in net reductions in emissions of most pollutants when avoided emissions from energy generation are taken into account. Energy generation from waste management generally gives rise to reductions in overall emissions of sulphur dioxide and particulates, when emissions from power stations are taken into account. Emissions of other pollutants from waste management are likely to give rise to no significant change or a net increase in emissions.

The information presented in Chapter 2 and Appendix 3 of this report could be used to assist in more detailed calculations of avoided emissions.

The information presented throughout this report could be used in the development of life-cycle assessment models for waste management. The Environment Agency's "Waste Integrated Systems for Recovery and Disposal" (WISARD) system is one such model, which was first published in 1999. The information on emissions contained in Chapter 2 could be used to feed into updates of WISARD and the further development of other such models.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	tification of the onmental equences of sions Context for Quantified Environmental Risks, 8 Public Perception Issue		d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation	ons for

Comparison between waste management and other sources of emissions to air

The following comparisons can be drawn between estimated emissions to air from treatment and disposal of MSW and other sources of air pollutants.

- Treatment and disposal of MSW accounts for a small proportion of UK emissions of oxides of nitrogen, sulphur dioxide and particulate matter (less than 1% in each case; data for this comparison of moderate quality).
- Treatment and disposal of MSW accounts for a small proportion of UK carbon dioxide emissions (approximately 2.5%), but a significant proportion of UK methane emissions (approximately 27%; data for these comparisons of moderate quality).
- Treatment and disposal of MSW accounts for less than 0.02% of UK emissions of volatile organic compounds (data for this comparison of poor quality)
- Emissions of dioxins and furans from treatment and disposal of MSW account for approximately 1% of the UK total (data for this comparison of poor quality). At present, MSW-related emissions of dioxins and furans are emitted approximately equally from landfill and incineration. MSW accounts for a very small proportion of emissions of polycyclic biphenyls, a closely related group of compounds.
- Emissions of metals from treatment and disposal of MSW are a small proportion of the UK total. Emissions of cadmium from MSW account for 10% of the UK total, mainly emitted from landfill (data for this comparison of poor quality).

7.2.2 Solid residues

The existence of less information on solid residues compared to emissions to air should not be taken to imply that solid residues are of less concern. Instead, it highlights a need for further research in the areas discussed below.

Solid residues from the treatment of MSW are handled in three main ways:

• Disposal to landfill

Any biodegradable component of the residue will degrade within the landfill. Its degradation will generate landfill gas and landfill leachate, which may subsequently be emitted to air, sewer or groundwater. These emissions have not been quantified in this study, although we have quantified emissions from the disposal of untreated MSW to landfill.

Land spreading

Spreading of compost or digestate from anaerobic digestion to land may be considered an emission from MSW treatment to land. The effect of substances contained in materials spread to land depends on the quantity and availability of any such substances. These are controlled through the application of standards such as British Standard BS PAS 100. This sets limits for human pathogens; potentially toxic elements (e.g. heavy metals); physical contaminants (e.g. glass, metal and plastic); substances toxic to plants; and weeds

Re-use

Some ash and char residues from thermal treatment processes can be re-used. Again, the potential exists for trace constituents of these substances to be leached out and potentially impact on receptors. These emissions have not



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	ed Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation future work	ons for

been quantified in this study, although field evidence (see Chapter 6) suggests that emissions to land, groundwater or surface waters in this way could potentially be significant.

Landfill does not give rise to an emission to land in this way. Information on emissions to land from mass-burn incineration is generally of good quality, following a recent Environment Agency study. Information on emissions to land from other processes is of moderate or poor quality, or is not known. Because land spreading represents a possible pathway for public exposure to contaminants, further research should concentrate on the quantity and composition of residues from composting, MBT and anaerobic digestion of MSW which are spread to land. Further work should also be carried out on the composition of ash arising from incineration of pre-sorted wastes.

MBT results in the greatest mass of solid residue per tonne processed (60% of the mass of MSW processed). This is to be expected because MBT is an intermediate step in waste management. Composting also gives rise to a significant mass of solid residue (50% of the mass processed). Residues from composting can provide a benefit when used to improve soil structure, and so this quantity of solid residue should not necessarily be viewed as a disadvantage of composting.

Mass burn incineration gives rise to an intermediate quantity of solid residue (bottom ash). Currently, in the UK, about one third of this is re-used, and two-thirds sent to landfill.

Pyrolysis/gasification gives rise to a relatively low quantity of solid residue per tonne processed, about half the quantity produced by mass burn incineration. All waste combustion processes use some form of air pollution control system to remove acids from the exhaust gases. The residues from these air pollution control systems are strongly alkaline, which means that they need to be disposed of as a special waste.

Emissions of dioxins and furans in solid residues per tonne of waste processed are greatest for mass burn incineration. This may reflect the presence of dioxins and furans in the unsorted feedstock to MSW incineration processes. Also, while steps are taken to minimise the formation of dioxins and furans in the combustion process, a low level of dioxin formation will nevertheless take place. The primary fate for dioxins formed in this way is in the air pollution control residues.

The metals content of composted MSW is generally lower than that of ash residues from combustion processes. No clear pattern of levels of metals between different combustion processes emerges.

Materials Recycling Facilities provide an opportunity for materials in the waste stream to be recycled. Reprocessing materials in this way could result in increases or decreases in solid residues from processes remote from the MRF itself. These complex issues lie outside the scope of this report, which focuses on the potential emissions and effects associated with the facility itself.

A discussion of possible emissions under non-standard operating conditions is given in Section 7.2.1 above.

7.2.3 Emissions to surface water/groundwater/sewer

There is in general a lack of information on emissions to surface water, groundwater and sewer which would enable different waste management options to be compared, compared to the data available to assess emissions to air. This is



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Iss	d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation future work	ons for

not necessarily problematic for operating and regulating individual facilities, as information is in general gathered for substances relevant to each individual process type. However, it does inhibit comparison between different policy options, and areas where further research would be helpful are highlighted below.

In view of the lack of data at present, we recommend that the quantity, composition and control of liquid effluent from composting, MBT and anaerobic digestion of MSW should be systematically investigated.

Emissions from landfill to surface water, groundwater and sewer have been studied and controlled for over 20 years. Landfill site operators have since 1989 undertaken a risk assessment to ensure that emissions do not pose an unacceptable risk to groundwater. In this report, we have estimated the quantity and composition of emissions from UK landfills to groundwater. Although these estimates are necessarily of poor or moderate quality, this should not be taken to imply that there may be a risk to health from these emissions. The existence of site-specific risk assessments; controls on trace contaminants in foods and drinking water; and treatment of drinking water all serve to control and minimise the risk to health from this pathway.

Again, emissions from landfill differ from other processes in terms of the timescale over which emissions take place. Emissions from processes other than landfill occur at approximately at the same time as the treatment of the waste. In contrast, emissions from landfill take place over a much longer timescale. Maintenance of the management and regulation of these emissions over the period of decades following closure while emissions are at a significant level is important, and financial provision is made to enable the management of closed landfills to be properly financed.

So far as comparative data exist, composting facilities appear to give rise to greater emissions of metals to sewer than landfills. However, composting facilities do not all give rise to a liquid effluent. The nitrogen loading of effluent discharged to sewer from MBT is higher than that of anaerobic digestion or landfill.

Materials Recycling Facilities provide an opportunity for materials in the waste stream to be recycled. Reprocessing materials in this way could result in increases or decreases in emissions to surface water, groundwater or sewer from processes remote from the MRF itself. These complex issues lie outside the scope of this report, which focuses on the potential emissions and effects associated with the facility itself.

A discussion of possible emissions under non-standard operating conditions is given in Section 7.2.1 above.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quar Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation	ons for

7.3 Health Effects

7.3.1 Epidemiological studies

We found no consistent evidence for significantly elevated levels of ill-health in populations potentially affected by emissions from MSW incineration.

We examined the observation that certain birth defects occur at slightly higher rates in people living close to landfills. The available information does not allow us to say whether the landfills cause or contribute to this apparent clustering of birth defects. One study also indicated that there may be increases in some acute health effects in populations living near composting facilities.

Other than the study of birth defects discussed below, we found no consistent evidence that people living close to landfill sites accepting MSW suffered worse health than people living further away from such sites. In particular, we found that the weight of evidence is against any increased incidence of cancers in people living near to landfill sites.

One recent statistical study (Elliott et al., 2001) has developed dose-response functions for an observed association between the incidence of birth defects and living within 2 km of open and closed landfill sites. The birth defects studied were neural tube defects; cardiovascular defects; hypospadias and epispadias; and abdominal wall defects (including gastroschisis and exomphalos; a glossary of these terms is provided at the start of Chapter 3 of this report).

The data derived from this study is subject to a number of significant limitations.

- The study was not able to state whether the observed increment was due to exposure to emissions from the landfill, or to some other cause.
- The observations may be the result of residual confounding (possibly due to misclassifications of socio-economic status) rather than a true reflection of an excess of disease.
- There is uncertainty in some cases to the nature of waste that was being disposed, and it is not clear whether the sites referred to as "non-special waste" sites may have received some hazardous wastes in addition to municipal refuse.
- For sites which opened during the study period, some of the outcomes considered were at a lower rate after the sites opened than before it opened. This indicates that factors other than the landfill sites may be the cause of the observed increases.
- The relatively small scale of incremental health risks identified in this study suggest that it is less likely that the reported effects are in fact caused by exposure to emissions from the landfill sites studied.

We recommend further work to determine the true significance of these findings. Progress in this area will depend on getting better information on residents' exposures to toxic substances. This will help to show whether a causal connection between landfill sites and human health is plausible. If a plausible source, pathway and dose-response association is identified, then appropriate measures can be designed to deal with the source and/or the effect. These measures may be



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Iss	d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation	ons for

implemented at the operational, regulatory and/or policy level. If found to be caused by the landfill, these impacts might be expected to arise over a longer period than the period of disposal of waste – as a worst case, approximately 20-30 years after disposal of waste.

A separate study identified dose-response functions for an observed association between certain acute health effects and living close to a commercial scale composting site. In view of the small number of composting facilities in the UK, and the likelihood that few people will reside within 400 metres of these facilities, any effects are likely to be small. There is likely to be an increased use of small-scale local composting facilities in the future, driven by the need to divert biodegradable wastes away from landfill. This would increase the need to investigate the local health consequences of composting.

When considering the health effects of an individual facility, it is important to take account of the local circumstances and any evidence of sensitivity of local residents to the health effects of concern.

7.3.2 Health effects of emissions to air

We used dispersion modelling to investigate the health effects of emissions to air from treating and disposing of MSW. We examined only airborne exposure pathways which could be quantified. We were able to examine the main pollutants of concern emitted from processes involving combustion.

This leaves three possible areas which we were not able to investigate by considering the possible health effects of emissions from waste management facilities. Firstly, not all combustion gas emissions could be quantified – in particular, we found no data on emissions of particulates from MBT or anaerobic digestion of MSW. Secondly, health effects may arise from the release of substances other than combustion gases to air. This is a particular concern for open windrow composting, but may also be relevant to landfill and MBT, and possible other facilities, particularly if wastes are not dealt with promptly.

Thirdly, we were not able to investigate the possible health effects arising from exposure to substances released to groundwater, surface water, sewer, or to land. This is because exposure via these routes is highly site-specific. Emissions to these media (e.g. to surface water or groundwater) are almost universally treated before consumption – in particular, drinking water is treated before supply and has to comply with strict quality standards. Products spread to land such as compost are now also subject to specific control standards. This reduces the likely significance of these pathways in respect of their potential impact on human health. Additionally, the health effects associated with exposure to dioxins and furans from thermal processes were found to have no significant adverse health effects. Nevertheless, a study of public exposures to substances via routes other than airborne exposure would be valuable.

As noted above, there is a lack of reliable data on emissions to air from composting facilities. The epidemiological research referred to above highlights potential effects on health outcomes such as minor acute symptoms, medication use and bronchitis. This suggests that a pathway for adverse health effects may exist, possibly linked to emissions of particulates or micro-organisms. Further work would be useful to investigate the plausibility of this potential pathway for health effects in local populations.

The atmospheric dispersion modelling study indicates that emissions to air from MSW management are not likely to give rise to significant increases in the adverse



Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Envi Con: Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	ed Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation future work	ons for

health effects studied. On a national scale, approximately 5 hospital admissions per year are estimated to arise due to emissions to air from treatment and disposal of MSW. Although this numerical estimate is of poor quality, it is indicative of a much lower incidence of hospital admissions compared to the 168,000 hospital admissions per year which result from accidents in the home.

Similarly, emissions from treatment and disposal of MSW are estimated to result in approximately one death being brought forward nationally every two years. This single statistical value is somewhat misleading, as the effect corresponds to a slight shortening of numerous individual lives. The numerical value is also of poor quality. The estimated incremental increase in cancers due to emissions to air from treatment and disposal of MSW is lower still. The effects described in this and the preceding paragraph would not be detectable in individuals.

This information is appropriate to typical facilities which may be operated in the UK. Any assessment of an individual facility would need to take account of the local circumstances and any known sensitivity to the emissions studied in this chapter.

These estimates of a low level of extra illness and deaths brought forward are derived from information of poor or moderate quality. Nevertheless, this information could be used to support the development of life-cycle assessment models such as the Environment Agency's WISARD system. Because the estimated health effects are low when compared with those for other major public health concerns, we suggest that new research should seek as a first priority to evaluate health risks which cannot be studied in this way.

Useful areas of research would include an investigation of the health effects of emissions from composting of MSW, and an investigation of the potential exposure to contaminants following re-use of MSW incinerator ash under conditions specific to the UK. We understand that investigations of re-use of incinerator ash are being carried out by the Waste Resources Action Programme (WRAP) and on behalf of some local authorities. This research may need to be co-ordinated to ensure that the information is as useful as possible for understanding and managing any health risks.

7.3.3 Health effects of emissions to surface water/groundwater/sewer/land

As noted in Section 7.2.2, we were not able to investigate the possible health effects arising from exposure to substances released to groundwater, surface water, sewer, or to land. Although adverse health effects due to exposure via these pathways are unlikely, a study of public exposures to substances via routes other than airborne exposure would be valuable.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Iss	d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation	ons for

7.4 Environmental Effects

Convincing national or international literature evidence for adverse environmental effects from managing MSW is very limited. Extensive recent research has investigated aspects of waste management under the direct control of waste management facility operators such as landfill gas composition and control, and the emissions from waste incinerators. Less attention has been paid to investigating whether and by what means waste management facilities could affect the environment in practice (other than the human health issues discussed above).

The issue of greatest concern relates to emissions of greenhouse gases. Emissions of methane from landfill of MSW are a particular concern, together with emissions of carbon dioxide from all MSW management facilities. Emissions of methane from commercial-scale composting of MSW may also be significant. We expect that current changes in the design and operation of landfill will significantly reduce these emissions.

Waste incinerators contribute to local air pollution. This contribution, however, is usually a small proportion of existing background levels which is not detectable through environmental monitoring (for example, by comparing upwind and downwind levels of airborne pollutants or substances deposited to land). In some cases, waste incinerator facilities may make a more detectable contribution to air pollution. Because current MSW incinerators are located predominantly in urban areas, effects on air quality are likely to be so small as to be undetectable in practice.

Emissions to air from MSW treatment and disposal also contribute to secondary effects on a regional scale, such as the generation of ozone and secondary particulates. Emissions from MSW facilities account at present for less than 1% of UK emissions of the primary pollutants such as oxides of nitrogen and sulphur dioxide. MSW processing probably contributes on a similar scale to secondary pollution. The effects are not likely to be significant in the context of other sources of emissions to air.

The effects of MSW landfilling on groundwater quality has been studied at individual sites. Even at older sites which are not engineered to current standards, significant adverse effects on groundwater quality are rare. The current generation of landfills is subject to a much more rigorous and demanding regulatory regime, which should further reduce the potential for adverse effects.

Landfills probably account for between 10% and 25% of all odour complaints to local authorities. Landfills and other waste management operations account for a smaller minority of noise complaints.

When considering the environmental effects of an individual facility, it is important to take account of the local environmental conditions, and any evidence of sensitivity to the potential environmental effects of the waste management facility.

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Quai Envi Cons Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation	ons for

7.5 Recommendations for future work

We have identified outstanding information needs, and we recommend research to address these needs as follows. The high priority recommendations are those which could be most readily carried out, and which would give information of greatest value.

High priority recommendations

- A field study of population exposure to substances emitted from landfill sites, paying particular attention to any substances which could potentially give rise to an increased incidence in the adverse birth outcomes discussed above. This would help to establish whether the observed association could be caused by exposure to landfill emissions. If a cause and effect linkage were to be identified, measures might be specified to reduce the significance of the effect.
- A study to characterise emissions of, and exposure to, bioaerosols and microorganisms released from all kinds of waste management facilities, focusing in particular on those with no combustion component. The following species should be considered: fungi, bacteria, actinomycetes, endotoxin, mycotoxins, and glucans
- A study to characterise and quantify emissions of particulates, micro-organisms, VOCs and methane from in-vessel and/or windrow composting of MSW This is a significant area of uncertainty at present, and could become more important if composting of MSW becomes more widespread.
- A study to characterise and quantify emissions to air, emissions to sewer, and solid residues from MBT, anaerobic digestion of MSW and any other techniques likely to play a role in treatment and disposal of MSW in the future. This should also characterise and quantify liquid effluent from composting of MSW *This is likely to become more significant if MBT and anaerobic digestion of MSW play an increasingly important role in the future.*

Other recommendations

- The preparation and publication in the peer reviewed literature of good quality information on emissions from all processes to all media This literature review evaluates data quality and notes the source of information, but cannot go beyond this. This is a particular concern for information on emissions which is published via routes which are not formally peer reviewed. Some information is published by reputable governmental or regulatory bodies, which provides some confidence in the data quality. Some data is published without independent checking of data quality.
- A study to investigate whether an increase in composting of MSW could have a significant adverse effect on health
 A study could be carried out using the dose-response functions provided in this report, drawing on information from the preceding research recommendations. While these effects are unlikely to be significant on a national scale at present, they could become more important if composting of MSW becomes more widespread.
- A study of emissions from and potential health/environmental effects of waste management facilities under non-standard operating conditions

Introduction	Review of Information on Emissions	Review of Epidemiological Research	Quantification of Health Consequences of Emissions	Qua Envi Con: Emis	ntification of the ronmental sequences of ssions	Context for Quantifie Environmental Risks Public Perception Is	d Health and , & Review of sues	Conclusions
Introduction	Emissions		Health Effects		Environmental Effe	ects	Recommendation	ons for

This should develop theoretical estimates of emissions under non-standard operating conditions, with evaluation against field data obtained during non-standard conditions. This information could then be used to evaluate potential health or environmental effects.

- Collation via the pollution inventory or otherwise of discharges from landfill to sewer, surface water or groundwater to verify the data set out in this report.
- A more detailed prediction of the likely health and environmental effects of the range of options for future integrated waste management in the UK. This could also consider the effects of the historic practice of co-disposal of special wastes with MSW. Although this practice will cease from 2004, emissions from landfills where this has taken place will continue for periods of at least 20 years
- A study to review research into the emissions, health effects and environmental effects of recycling processes This would enable the effects of recycling processes to be considered alongside the effects of waste disposal operations set out in this report. This would permit more informed judgements to be taken about the most appropriate ways of managing waste
- A study to evaluate possible new and evolving techniques for MSW management to ensure that a full range of information is available to those working in the waste management field
- A study to improve the estimated health effects associated with emissions to air from landfill set out in this report. These estimates are currently of poor quality

APPENDIX 1	APPENDIX 2	APPENDIX 3	APPENDIX 4
Bibliography			

APPENDIX 1 : BIBLIOGRAPHY

APPENDIX 1	APPENDIX 2	APPENDIX 3	APPENDIX 4	
Bibliography				

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Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Un- segregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

APPENDIX 2 : LITERATURE REVIEW OF THE ENVIRONMENTAL EFFECTS OF WASTE MANAGEMENT OPERATIONS

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Un- segregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.1 Introduction

This appendix relates to Chapter 5, and reviews and discusses in detail the environmental effects of waste management operations. As discussed in Chapter 5 the information available on environmental effects was found to be limited in coverage and quality.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Un- segregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.2 Environmental Statements

Virtually all UK waste management facilities are now going through a process of environmental impact assessment. This means that new or extended facilities are only being implemented if the projected impacts have been found to be acceptable in their local context. Waste management processes are regulated to ensure that they conform with the performance identified at the environmental impact assessment stage. Nevertheless, in common with other industrial processes, waste management facilities will produce impacts, albeit at an acceptable level in their local context, and these are assessed in this chapter based on the information in the available references.

The environmental effects identified in Environmental Statements for waste management facilities are set out in Table A2.1.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportatio n	Anaerobic Digestion with Energy Recovery	Soil Acidification

 Table A2.1
 Environmental effects identified in Environmental Statements for waste management facilities

Name of Facility	Type of Facility	Location	Date of ES	lssues considered	Air Quality	Water Environment	Noise and vibration	Other
Guernsey Integrated Waste Management Facility	EfW Facility, 70KT, MRF and CA site	Longue Hougue, Guernsey.	2001	Air Quality, Landscape and Visual impact, Natural Heritage, Cultural Heritage, Water environment, Management of Residues Community Effects, Land Use, Traffic and Transport, Noise and Vibration.	Using a 20m effective stack height and semi dry emission abatement equipment, modelling predicts that ground level concentrations will exceed 2% of the Environmental Assessment Level on the following criteria: hydrogen chloride, dioxins and furans, VOC (expressed as benzene), cadmium, sulphur dioxide and arsenic. On a 30m effective stack modelling predicts that ground level concentrations will exceed 2% of the Environmental Assessment Level on cadmium as an annual average and VOC as an annual average. Maximum ground impact from stack emissions (from plume prediction exercise) between 150 and 270 metres from the facility. Levels assumed are the UK AQS and Environmental Assessment Levels. For all EALs to be met a stack height of 43 to 49 metres for a 25 metre high building required. Odour not a significant problem as air drawn into plant.	Water discharges to be treated to levels as required by EC law. No data on levels emitted only that 'waste from municipal waste incinerator plant have been shown to be contaminated with heavy metals and inorganic salts and to have high temperatures and high alkalinity or acidity. Scrubber waters can have a very low pH and contain metal hydroxides.' Issue raised over leaching of bottom ash stored on site – mostly in the form of sulphates and chlorides with Copper being the most leachable element. Mitigation measures for any impacts.	Predicted levels as nearest residential property at 38.8dB (from Static plant) 37.4dB (from mobile plant), 47.9dB in Industrial area (from static plant), (44.9dB from mobile plant). Construction Noise in excess of these levels for ~28 months. Vibration other than during construction should not be an issue due to anti-vibration mounts for all vibrating machinery.	For residue management: Dutch leachability limits used. Assumed dry or semi dry APC system. Bottom Ash ~20,370 T for 70,000 T input, ~3150 T for Flyash / APC residue (to be stabilised or encapsulated prior to landfill).

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportatio n	Anaerobic Digestion with Energy Recovery	Soil Acidification

Name of Facility	Type of Facility	Location	Date of ES	lssues considered	Air Quality	Water Environment	Noise and vibration	Other
Integrated Waste Management Facility	EfW Facility, Oscillating Kiln, 55KT/a, Composting plant, Bulking up plant	Stallingborough, NE Lincolnshire.	2000	Noise, Archaeological Impacts, Land Use, Water Environment, Transport, Community, Landscape and Visual Impacts.	Lead at 0.5µgm ⁻³ is the determining factor for stack height which is required to be 55 metres.	Contained in technical Appendix - unavailable.	No properties within 1km - so only minor mitigation measures required.	Bottom Ash predicted at ~25% of waste input by weight (14100T/a). Flyash and APC residue ~4% by weight (1740T/a).
Vine Street MRF and Waste to Energy Development	MRF and 150KT / a EfW facility	Vine Street, Huddersfield, West Yorkshire.	1999	Air Quality, Noise, Traffic, Visual Amenity, Water Impacts, Socio- economic, Ecology, Architectural / Cultural Issues.	93m stack, semi dry APC system with DeNox. Stack height set by existing structure, and found to be acceptable based on air modelling study. NO ₂ is key impact, as facility located close to a trunk road. Technical Appendix missing.	MRF does not use significant amounts of water. Water abstraction for the EfW process uses 100,000 cubic metres of water, rainwater also collected and used.	Operational Noise varies between 39 and 47dB in daytime.	
Kennett Hall Farm – revised and updated ES	153kT/a Landfill (for 21 years)	Kennet Hall Farm , Cambridgeshire.	2002	Landscape, Surface Water Management, Agriculture and Soils, Ecology, Air Quality / Odour, Noise, Access / Traffic, Cultural Heritage,	Dust and Litter – large dust particles mostly deposited within 100m of sources and 200 – 500m for intermediate sized particles. Fine dusts, which make up a small proportion of workings can travel up to 1km from sources, but have a low deposition rate and so are 'unlikely to create any amenity effect'.	Balancing pond for surface water, with 'ecological features'. Low permeability clay and membrane liner.	At nearest receptor, minimum distance of 30m, the calculated noise level of 70dB (A) Leq, which is 15dB(A) above the reference level (however it will be unoccupied as it is owned by the operator). At a receptor min.130m from site there would be noise levels 3dB (A) above the reference level. Amelioration measures can be used to minimise / negate this.	42 HGV vehicle movements per day (in addition to 41 movements at other existing part of the site) for operational purposes and 44 – 58 movements per day for construction and restoration purposes of the site. These movements are spread amongst the access routes to the site to minimise impact.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4 Anaerobic Waste Digestion		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportatio n	Anaerobic Digestion with Energy Recovery	Soil Acidification

Name of Facility	Type of Facility	Location	Date of ES	lssues considered	Air Quality	Water Environment	Noise and vibration	Other
Residual Landfill Ballyguyroe	145kT/a Landfill (for 10 years)	Ballyguyroe, County Cork, Ireland	2001	Climate, Air Quality, Noise, Soils / Geology, Surface Water, Groundwater / Hydrogeology, Ecology, Human Beings, Roads and Traffic, Landscape and Visual, Cultural, Material Assets.	Dust may be generated in significant quantities during the construction of the site unless properly managed. Aerosols can occur where leachate is aerated or recirculated, however on this site recirculation will take place beneath the cap and it is not proposed to treat or aerate leachate on site and it will be tankered off-site for treatment. Faint odours detected at 50 and 100m from site with no odour detected at 200m from site. 'Although odour can be detected, many compounds are present at a concentration that poses no health risk to the receiver'.	Leachate will be removed from the landfill and tankered off-site for treatment. Surface water will be treated to remove colloidal clay particles prior to discharge at the local drainage network. Low permeability clay and membrane liner. Flows from hardstanding areas to pass through an oil interceptor prior to discharge to the surface water siltation pond.	At one receptor a maximum predicted increase of noise would be 3-4dB(A).	Flaring and utilisation of Landfill Gas to reduce global warming impacts. Maximum of 170 daily HGV movements to the site during construction phase, with approximately 100 daily HGV movements for typical operational activities.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4 Waste Digestion Transportatio with		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportatio n	Anaerobic Digestion with Energy Recovery	Soil Acidification

Name of Facility	Type of Facility	Location	Date of ES	lssues considered	Air Quality	Water Environment	Noise and vibration	Other
Slyfield Integrated Waste Management Centre	EfW plant, 225,000T/a, MRF and CA site	Slyfield Industrial Estate, Guildford, Surrey	2000	Air Quality, Culture and Heritage, Ecology, Geo- technical, Landscape and Visual Impact, Socio- economic Considerations, Traffic and Transportation, Waste Disposal and Water Resources, Health Impacts.	CO ₂ , NO _x , SO _x , CO, Particulates (including PM ₁₀), HCl, HF, Hydrocarbons, PCDD/F and metals were assessed. Based on a 70m stack height, maximum process contributions to short term peaks of pollutant were examined. These were all significantly below environmental limits and air quality standards, with the closest emission to the standards being NOx at 48μ g.m ⁻³ which is 24% of the 200 μ g.m ⁻³ standard. Only Cadmium exceeded 3% of the Long Term Criteria Standards, with a worst case assessment of 12% of the emissions standard. Cumulative emissions from plant operations including traffic and background emissions show a cadmium impact of <28% of the relevant Environmental / Air Quality limit, NO ₂ at 33 – 41% and SO ₂ at <23%.	Perimeter bund to remove any flood risk to site and resultant potential pollution impact.	Noise from pile driving in construction is likely to be 8-9dB(A) above existing daytime levels at nearest dwellings. Operational noise would not be in excess of background levels.	286 daily vehicle movements to and from site. Health impact emissions were assessed. Dioxin exposure, based on a maximum exposed individual for 70 years. Through an atmospheric route this would give a total lifetime dose of 0.57ng, which equates to 0.015pg / day which is 0.007 – 0.03% of the WHO Tolerable Daily Intake (TDI). Through ingestion the predicted worst case intake would be 0.026% of the TDI. Similar health studies were undertaken for trace metal uptake which were at least three orders of magnitude below reference dose criteria. Facilities for the recycling of bottom ash (67,500T) on site, flyash and APC residues (13,500T) sent off-site to Landfill.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4 Waste Digestion Transportatio with Energy Porceptor		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportatio n	Anaerobic Digestion with Energy Recovery	Soil Acidification

Name of Facility	Type of Facility	Location	Date of ES	Issues considered	Air Quality	Water Environment	Noise and vibration	Other
Wrexham Resource Recovery Centre	MRF, RDF and Tunnel Composting Integrated Facility	Wrexham, Wales	2001	Water Impacts, Air Quality, Noise and Vibration, Visual Impact, Traffic, Ecology, Impact on the Ground.	The pollution contribution of the composting, RDF and MRF plant were predicted. Emissions were set against the relevant Environmental Assessment Levels or National Air Quality Objectives. Particulates, NO ₂ , SO ₂ , CO, HCI, HF, Hg, Cd, metals, TOC and Dioxins were assessed. The most significant was that for NO ₂ , however this was only 12.9% of the limit. When the process contribution was added to background concentrations the predicted environmental concentrations for particulates was 48% of the guideline, cadmium 20%, other metals 27% and NOx 28% of their respective guidelines. Overall development likely to adversely affect air quality to a minor degree.	Systems are essentially 'enclosed' and so any discharge from processes (e.g. composting) can be controlled. Run off during construction discharged to sewer.	Site operations and construction traffic predicted to have a minor noise impact. There will be no plant vibration issues.	Construction traffic predicted to have a minor – medium impact during this phase. Otherwise minor impacts.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4 Waste Digestion Transportatio n Energy		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportatio n	Anaerobic Digestion with Energy Recovery	Soil Acidification

Name of Facility	Type of Facility	Location	Date of ES	lssues considered	Air Quality	Water Environment	Noise and vibration	Other
Marchwood Energy Recovery Facility	165KT/a EfW plant	Marchwood Industrial Park, Marchwood, Hampshire.	2000	Land Use, Traffic, Noise, Water, Geology and Soils, Cooling Water Abstraction and Discharge, Landscape and Visual Amenity, Air Quality, Ecology, Archaeology and Cultural Heritage	Minor dust impacts of construction. Predicted operational impacts of NOx, SO ₂ , particulates, CO, HCI, metals and dioxins and furans, were modelled using a 65m stack height. The maximum NO ₂ level predicted was 52µg m ⁻³ which represents ~18% of the EPAQS guideline. Ground level concentrations of NOx represents 27 - 29µg m ⁻³ (on the 99.8 T ^h percentile) compared to 23 – 50µg m ⁻³ background levels. The emissions from the proposed plant will have a small impact on the local air quality and ground level concentrations are unlikely to lead to breaches of ambient air quality standards and guidelines or cause a risk to health. The lifetime carcinogenic risk arising from inhalation and ingestion of trace metals or dioxins / furans is negligible.	Water abstraction for cooling and discharge was assessed and considered to not have a significant impact on the water and aquatic life environment. Otherwise zero process water discharge.	Construction Noise predicted at 7dB(A) above the ambient at some residential properties (up to a 'moderate' impact), whilst at the NW corner of the industrial park an increase of 16dB(A) (substantial impact) is predicted and suitable mitigation advice is provided. There would be no noise impact from the operational facility if the boundary noise limit of 53dB(A) LAeq, 1hr is imposed and the acoustic performances specified are met.	Traffic to the facility will represent a maximum of 25 vehicles per hour travelling to and from the site and will account for about 13% of traffic on the supporting road during the peak (midday) period. This represents between 2% and 7% at peak travel times (am and pm). Odours reduced through negative pressure of facility and good housekeeping. Bottom ash recycled on site.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre-sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Com- posting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportatio n	Anaerobic Digestion with Energy Recovery	Soil Acidification

Name of Facility	Type of Facility	Location	Date of ES	lssues considered	Air Quality	Water Environment	Noise and vibration	Other
Copyhold Energy from Waste Plant	225KT/a EfW Facility	Copyhold Works, Nuffield Road, Redhill, Surrey.	2000	Air Quality, Noise and Vibration, Traffic, Landscape and Visual Impacts, Ecology, Water Resources, Socio- Economic, Other Environmental Impacts.	NO ₂ , SO ₂ , particulate matter, CO, gaseous or vaporous organic substances, HCI, HF, metals and dioxins/furans assessed. The long term impact of any of the pollutants assessed does not exceed 3.8% of the 'Long Term mean air quality standards'. Over short term impacts all pollutants measured are all less than 20% of the relevant standard, with only NO ₂ around this level and the others less than 6.7% of their relevant standard.	All surface and foul water directed to sewer.	Noise reduction measures proposed to ensure acceptable levels of noise at noise sensitive developments.	60 daily HGV movements associated with construction phase. 60 loads of waste delivered to the facility each working day (32 refuse collection vehicles, 28 tipper trucks), plus 2 additional HGVs on miscellaneous deliveries. Using IEMA guidance no material adverse traffic related impacts are predicted in respect of the local road network.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.3 Unsegregated Incineration with Energy Recovery

Noise

Number of relevant documents: 3

Summary of findings:

The ODPM/Enviros (2003) draft report suggests that noise at incineration facilities may be associated with vehicle manoeuvring, induced draft fans, air cooled condensers and steam release valves/pipework. However, noise issues tend to be associated primarily with poorly maintained plant. Typical noise limits are 45— 55dB(A) (daytime) and 35-45dB(A) (night-time) as for other similar industrial process. Processes occurring within a building tend to control noise to acceptable levels.

Enviros (2003) report for Norfolk County Council suggests that noise may be contained using soundproofing in the walls of the plant.

The Environment Agency (2002f) suggests that the enclosed nature of the operation limits the potential for noise nuisance at incineration plants but that noise sources may include fans, and gas cleaning systems. However, noise is relatively easily controlled via soundproofing and bunds etc.

Comment on data quality:

Information is limited and qualitative.

Potential effects:

Like any industrial process, incineration may cause noise. However, the enclosure of operations within a building usually prevents any significant nuisance. Noise may also be attributed to associated vehicle movements. Sensitive site location, bunding, fencing and the use of smart reversing alarms (i.e. those that limit their output to 5dB(A) above background) may assist in controlling noise from such sources.

Odour/dust

Number of relevant documents: 3

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that odour should not be a problem at incineration plants unless waste is stored for long periods and allowed to decompose. Dust may be derived both from vehicle movements/unloading etc. and from storage/ transport of ash. The use of negative pressure within the building (and usually using the internal air as combustion air) controls dust and odour emissions. Vehicle sheeting and covered storage of ash also assists in dust control.

Enviros 2003 report to Norfolk County Council and McLanaghan (2002) suggest that odours and dust may be controlled by the use of negative pressure within the building and that odour at the perimeter of the plant is likely to be low.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

The Environment Agency (2002f) suggests that the enclosed nature of the operation limits the potential for noise nuisance at RDF plants and that any odours which do arise are likely to derive from fugitive emissions or waste handling operations. However, good plant design, waste handling practices and good housekeeping can eliminate odour. Use of internal air for combustion assists in odour control.

Comment on data quality:

Information is limited and qualitative.

Potential effects:

Given enclosure within a building odours and dusts are unlikely to represent problems at incineration plant, especially if negative pressure and use of internal air for combustion is adopted and care is taken in the storage and handling/transfer of ash. However, good housekeeping, including, sheeting and washing of vehicles, short waste storage periods, water bowsing / road sweeping and appropriate storage of dusty materials, may also assist.

Fauna

Number of relevant documents: 5

Summary of findings:

The principal route by which incineration processes could affect fauna is via the deposition of substances emitted to air. The possibility also exists for soil contamination via leaching of substances from incinerator ash streams.

Lisk (1988) discussed a range of research papers including a project where elevated soil levels of dioxins (8-2400ppb or μ g/kg) were studied in relation to microbial activity, with little or no decline in activity. This suggests either low microbial toxicity or low bio-availability once in soils (Arthur and Frea 1987). Another study showed that earthworms living within contaminated soils exhibit a 5-fold increase in dioxin levels in tissues compared to the surrounding soils within 5-days, indicating that bioaccumulation occurs (Reineke and Ash 1984). The rate of release and hence deposition of substances released from incinerators has decreased considerably since this study was published (see Section 4), reducing the likely significance of this pathway.

In addition, several studies indicate that bioaccumulation and toxic reaction to dioxins have been observed in fish exposed to leachate from fly ash (which could arise in the event of an escape from an ash disposal site or from a release of contaminated quench water). This is supported by Fielder (1996) who documents the high level of bioaccumulation of dioxins in fatty tissues of animals (consistent with the findings of Her Majesty's Inspectorate of Pollution, 1996).

Travis and Blaylock (1995) discuss mercury dispersion from all anthropomorphic sources including MSW incineration, and subsequent bioaccumulation in fish, identifying a median background level of 350ppb, with a range of 20ppb to 1500ppb in freshwater species in the USA. The paper also indicates that up to 58% of human mercury uptake is via fish and 31% via vegetation. Based upon mercury's bioaccumulation potential it is logical to deduce that mercury is bioaccumulated up through food-chains. The paper suggests that 4% of anthropogenic mercury in the

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
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USA is derived from municipal waste incinerators, although data in Chapter 7 indicate that the proportion in the UK is now between 1-2%.

Perrodin et al (2002) reported a French study involving the exposure of various soil fauna, plants and aquatic organisms to both MSW bottom ash and solidified APC residue percolates/leachates in the lab and in the field based on 'real world' scenarios (e.g. use of bottom ash as a road embankment in a mountainous area and use of APC residues to construct a fire reservoir which then leaks). Exposure to MSW bottom ash percolates resulted in no change in soil bacteria numbers although community structure was slightly affected. However, the presence of *Agrostis* reduced this effect suggesting that plant cover can protect soil from the addition of percolate via aerial watering. No effect on earthworms was noted. Percolate was toxic to aquatic invertebrates, with different species showing different sensitivities. Toxicity was reduced after passing through the sub-stratum. Pollution affected reproduction rates, diversity/trophic equilibrium of communities, species richness and abundance, and modified the whole aquatic system. In relation to the APC percolate microbial numbers did not decrease but communities were altered. Earthworms reduced their litter consumption.

Hwang (1990) provides modelled results for theoretical concentrations of 2,3,7,8-TCDD in beef and fish (in a pond 4000m² and 5m deep or large river - 300m wide, 5m deep, flowing at 0.5m/s) 800m from stack emissions from a conceptual incinerator burning 3000 T of waste per day. This is larger than any UK facility, and the release rate of dioxins per tonne of waste processed is approximately twice the current UK value (see section on air quality). The results are as follows:

Beef = 2.2×10^{-1} ng/kg Fish (pond) = 1.6×10^{-2} ng/kg Fish (river) = 1.6×10^{-6} ng/kg

This is a theoretical study of a very large facility, which in common with most quantitative exposure calculations of this nature makes a number of worst-case assumptions. Actual levels would be expected to be much less than these values.

Concentrations in beef and fish due to fly ash disposal to land at 30m and 150m from the disposal site are given. However, no details are given as to the assumptions for these concentrations (e.g. the containment that is assumed to be provided).

Comment on data quality:

The data are primarily limited to effects due to dioxins and metals from atmospheric emissions or due to contact with ash. Studies are either site specific or relate to theoretical modelled situations which inherently include a range of assumptions. However, the Hwang (1990) study does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

Potential effects:

Heavy metals and dioxins/furans are known to bioaccumulate in animals (especially in fatty tissues and the liver). Where exposure is high, top predators may be expected to accumulate significant quantities of metals and dioxins. Mammals, in particular, may be especially prone to such toxins due to passing pollutants to their offspring via the placenta and in milk. However, given the results of some of the studies referenced below for soils and vegetation, incinerators appear to provide a

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small proportion of such pollutants to the environment, in comparison to other sources, and non-cumulative effects are unlikely. Other effects may include habitat loss due to incinerator construction, as at any other industrial facility.

Flora

Number of relevant documents: 8

Summary of findings:

The principal route by which incineration processes could affect flora is via the deposition of substances emitted to air. Emissions of acid gases or nutrient species to air could have significant impacts on sensitive habitat sites. These issues are managed via the identification of critical load functions for different habitat types, and assessment of actual deposition of acid gases or nutrient nitrogen species against these critical loads. Lichens are particularly sensitive to deposition of sulphur dioxide, but many habitats are sensitive to acidification and eutrophication due to the deposition of excessive acids and nutrients. Unless properly controlled, emissions from waste management facilities could make a significant for MSW incineration processes. The possibility also exists for soil contamination via leaching of substances from incinerator ash streams.

Lisk (1988) cited studies that demonstrated bioaccumulation in plants and accumulation in soils of metals emitted from incinerators. The review also illustrated that bioaccumulation appears to be species dependent. A study from Finland (Kukkonen and Raunemaa 1984) suggested that concentrations of 10 elements (bromine, calcium, chlorine, iron, nickel, lead, silicon, titanium, vanadium and zinc) in birch leaves demonstrated a strong inverse relationship with distance from an incinerator. This phenomenon was noted from only a few elements in grass samples from similar locations. Experiments undertaken with cabbage and barley (Wadge and Hutton 1986), grown in soils contaminated with high levels of fly ash (10-40%), demonstrated elevated uptake of metallic compounds. Indeed, crops grown on 20% ash amended soils contained 146 times the cadmium found in control plants. Giordano et al (1983) also found that Swiss chard, when grown in soil containing 30% fly ash showed uptake of lead and cadmium. However, phytotoxicity was due to the fly ash salt content. Mika et al (1985) found no phytotoxic effects around an incinerator ash disposal site adjacent to a freshwater wetland. Lisk also noted that incinerators could contribute to acid rain which could The rate of release and hence deposition of substances then affect plants. released from incinerators has decreased considerably since this study was published (see Section 4), reducing the likely significance of this pathway.

Perrodin et al (2002) indicated that exposure to MSW bottom ash percolates resulted in a reduction in biomass and percentage germination of *Agrostis* plants in the lab. In the field, there was some evidence of sodium/copper accumulation but no effect on aerial/root (a/r) ratio. In relation to the air pollution control (APC) residue percolate, *Agrostis* showed an increase in biomass and germination was not affected in the lab. In the field, root biomass increased and sodium/copper accumulation was noted.

In a series of related papers Domingo, Llobet and Nadal took soil and herbage samples around three incinerators in Catalonia in order to assess the level of contamination in relation to the incinerator emissions. These studies are summarised below in relation to vegetation contamination:

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Domingo (2000) measured concentrations of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in the vicinity of a municipal solid waste incinerator in Adria Del Besos, Barcelona, Spain, (which started operations in 1975 and took 300,000 tonnes of waste per annum) by analysis of soil and vegetation samples up to 1km away. The results were compared to levels measured the year before. PCDD/F concentrations in vegetation in 1998 ranged from 0.33 to 1.98ngl-TEQ/kg, with median and mean values of 0.58 and 0.70ngl-TEQ/kg respectively. In 1999, PCDD/F concentrations in vegetation ranged from 0.32 to 2.52 ngl-TEQ/kg, with median and mean values of 0.82 and 0.97 ngl-TEQ/kg respectively. During the 12 month period, PCDD/F levels increased in 17 of the 24 vegetation samples analysed.

In relation to the same incinerator Domingo et al (2002) considered the results of an adaptation to the stack in 1999, when acid gas and metal emission limit equipment, together with an active-carbon adsorption filter were installed. PCDD/F levels in vegetation were used as a suitable indicator of the atmospheric PCDD/F emissions over a short period of time. Results of a 2000 survey were as follows: PCDD/F concentrations ranged from 0.22 to 1.20 ngl-TEQ/kg (dry matter), with median and mean values of 0.57 and 0.61 ngl-TEQ/kg (dry matter) respectively. 2001 values were: PCDD/F concentrations ranged from 0.23 to 1.43 ngl-TEQ/kg (dry matter), with median and mean values of 0.58 and 0.66 ngl-TEQ/kg (dry matter) respectively. These results are not significantly different to those given in the previous paper. Although concentrations in herbage samples were comparable to those found in recent surveys in other parts of Catalonia, the data indicated that other emission sources of PCDD/Fs also have a significant environmental impact on the area under direct influence of the incinerator. The study concluded that, in areas subjected to the environmental influences of a modern incinerator, or an incinerator equipped with modern technologies that allow emissions of less than 0.1 ngI-TEQNm⁻³, efforts to reduce atmospheric levels of these pollutants should be focused on those PCDD/F emission sources which are currently quantitatively more important than municipal waste incinerators.

Domingo et al (2001) considered the levels of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in the vicinity of another municipal solid waste incinerator (which also started operations in 1975) in Montcada, Barcelona, Spain, by analysis of soil and herbage samples. Levels were surveyed in 1998 and the results compared to levels in 1996 and 1997. PCDD/F concentrations in herbage in 1998 ranged from 0.40 to 1.94 ngl-TEQ/kg (dry matter), with median and mean values of 0.86 and 0.95 ngl-TEQ/kg (dry matter) respectively. Comparison with 1996 and 1997 surveys showed that PCDD/F concentrations decreased substantially in herbage samples over the two years. 1996 results were: range = 1.07-3.05 ngl-TEQ/kg (dry matter) median = 1.89 ngl-TEQ/kg (dry matter), mean = 1.9 ngl-TEQ/kg (dry matter). 1997 results were: range = 0.75-1.95 ngl-TEQ/kg (dry matter), median = 1.27 ngl-TEQ/kg (dry matter), mean = 1.3 ngl-TEQ/kg (dry matter). Whilst the technical characteristics of the incinerator, as well as the total amount of MSW incinerated, remained unchanged, PCDD/F accumulation could be counteracted by a decrease in the atmospheric levels of these pollutants from other emissions sources in the area, which probably accounts for the decrease in concentrations in herbage samples.

Nadal et al (2002) measured congener concentrations of polychlorinated dibenzo-pdioxins (PCDDs) and dibenzofurans (PCDFs) in the area under direct influence of Montcada incinerator in Spain to determine whether the environmental levels of these pollutants were mainly due to emissions from the plant. Soil and herbage samples were collected near the MSWI before (1998) and after (2000) technical improvements to the incinerator (acid gas and metal emission limit equipment

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

together with an active-carbon adsorption filter) were installed. PCDD/PCDF congener profiles were determined and compared with those collected in a suburban area outside the zone of influence from any incinerator. In all herbage samples, OCDD, 1,2,3,4,6,7,8-HpCDD, OCDF and 1,2,3,4,6,7,8-HpCDF were the congeners showing the higher percentages of contribution. The results suggested that the incinerator was not the main source of environmental PCDD/F concentrations in the area, and that other emission sources seem to have a notable impact on the atmospheric levels of these pollutants.

Llobet et al (2002) measured concentrations of metals in the vicinity of a municipal solid waste incinerator in Tarragona, Spain, by analysis of soil and herbage samples. Part of a long term monitoring programme, this paper considered the results of an adaptation to the stack in 1997, when acid gas and metal emission limit equipment together with an active-carbon adsorption filter were installed. Arsenic, cadmium, chromium, lead, manganese, mercury, nickel and vanadium levels were measured and compared to the results of surveys in 1994 and 1997. In vegetation, only manganese levels showed a significant reduction, whilst increases were found in the concentrations of arsenic, mercury and nickel. The paper concludes that other metal emission sources in the area of study were masking the environmental improvements carried out in the MSWI.

Comment on data quality:

The data are primarily focussed on accumulation of metals and PCDDs/PCDFs in plant tissue (especially the Spanish studies) with little information available on the effects of such accumulation on plants. The main concerns are likely to relate to effects on the food chain rather than directly on vegetation.

Potential effects:

From the above research toxins may potentially accumulate in plant tissue. However, this tends to have little effect on the plants themselves and the effect of toxin release from incinerators is relatively minor compared to other sources. Other effects may include habitat loss due to site construction as at any other industrial facility.

Soils

Number of relevant documents: 6

Summary of findings:

The principal route by which incineration processes could affect soil quality is via the deposition of substances emitted to air. The possibility also exists for soil contamination via leaching of substances from incinerator ash streams.

Nouwen et al. (2001) undertook an investigation of dioxin emissions from 2 MSW incinerators and the impacts upon soil and foodstuffs grown in the soil. Dioxin congener distribution in the soil did not match with the soil congener pattern predicted from the emissions from the incinerator, indicating that dioxins from other sources predominate in the soil. The soil sampling was based upon 15 samples, obtained between 300m and 3.5km upwind and downwind of the plants. The average soil dioxin concentration was 10 ngTEQ/kg (dry matter), and the range was 1.5-27.2 ngTEQ/kg (dry matter). The variation in the soil dioxin concentrations did not bear a noticeable relationship to distance from the incinerator nor to the

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

direction relative to the prevailing winds. This indicates that the primary dioxin sources in the area were not the incinerators.

Hwang (1990) provides modelled results for theoretical deposition rates and soil concentrations of 2,3,7,8-TCDD 800m from a conceptual incinerator burning 3000 T of waste per day. This is larger than any UK facility, and the release rate of dioxins per tonne of waste processed is approximately twice the current UK value (see section on air quality). The results are as follows:

Deposition rate at $800m = 0.63 \text{ ng/m}^2/\text{yr}$

Soil concentrations at 800m = 0.54 ng/kg, consistent with the findings of Nouwen (2001).

In view of typical UK facility size and emission rates, actual levels are likely to be lower than these values. Soil concentrations due to fly ash disposal to land at 30m and 150m from the disposal site are given. However, no details are given as to the assumptions for these soil concentrations (e.g. the containment that is assumed to be provided).

In a series of related papers Domingo, Lobett and Nadal took soil and herbage samples around various incinerators in Catalonia in order to assess the level of contamination in relation to the incinerator emissions. These studies are summarised below in relation to soil contamination (see Flora section also):

Domingo et al (2000) measured concentrations of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in the vicinity of the Adria Del Besos incinerator, Barcelona, Spain, by analysis of soil and vegetation samples. The results were compared to levels measured one year before. PCDD/F concentrations in soil in 1998 ranged from 1.22 to 34.28ngl-TEQ/kg, with median and mean values of 9.06 and 12.24ngl-TEQ/kg (dry matter) respectively. In 1999, PCDD/F concentrations in soil ranged from 1.33 to 54.23ngl-TEQ/kg, with median and mean values of 11.85 and 14.41ngl-TEQ/kg respectively. During the 12 month period, PCDD/F levels increased in 16 of the 24 soil samples analysed.

Domingo et al (2001) considered the levels of polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) in the vicinity of the Montcada incinerator, Barcelona, Spain, by analysis of soil and herbage samples. Levels in 1998 were compared to levels in 1996 and 1997. In 1998 PCDD/F concentrations in soil ranged from 0.06 to 127ngl-TEQ/kg (dry matter), with median and mean values of 4.80 and 9.95ngl-TEQ /kg (dry matter) respectively. Comparison with 1996 and 1997 surveys showed that no significant differences in the concentrations of PCDD/F in soils were found. 1996 results were: range = 0.28-44.3ngl-TEQ /kg (dry matter), median = 3.52ngl-TEQ /kg (dry matter), mean = 6.91ngl-TEQ /kg (dry matter). 1997 results were 0.15-29.7 ngl-TEQ /kg (dry matter), median = 2.5 ngl-TEQ /kg (dry matter), mean = 4.48ngl-TEQ /kg (dry matter).

Nadal et al (2002) measured congener concentrations of polychlorinated dibenzo-pdioxins (PCDDs) and dibenzofurans (PCDFs) in the area under the direct influence of the Montcada incinerator to determine whether the environmental levels of these pollutants were mainly due to emissions from the plant. Soil and herbage samples were collected near the MSWI before (1998) and after (2000) when technical improvements to the MSWI and metal emission limit equipment together with an active-carbon adsorption filter were installed). PCDD/PCDF congener profiles were determined and compared with those collected in a suburban area outside the zone of influence from any incinerator. The congener profiles which were the main
	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

contributors in percentage terms in soil samples were OCDD and 1,2,3,4,6,7,8-HpCDD. The results suggested that the incinerator was not the main source of environmental PCDD/F concentrations in the area, and that other emission sources seem to have a notable impact on the atmospheric levels of these pollutants.

Llobet et al (2002) measured concentrations of metals in the vicinity of a municipal solid waste incinerator at Tarragano in Spain, by analysis of soil and herbage samples. Part of a long term monitoring programme, this paper considered the results of an adaptation to the stack in 1997, when acid gas and metal emission limit equipment together with an active-carbon adsorption filter were installed. Arsenic, cadmium, chromium, lead, manganese, mercury, nickel and vanadium levels were measured and compared to results of surveys in 1994 and 1997. The only significant changes in soil levels corresponded to decreases in cadmium and lead. The study concluded that other metal emission sources in the area of study were masking the environmental improvements carried out in the MSWI.

Lisk (1988) discussed a range of studies including one by Berlincioni and di Domenico (1987) who sampled soil up to 1km from an incinerator and found a maximum PCDD concentration of 70,000 mg/m² of soil surface. The compounds were not limited to the top 5cm, probably due to the effects of leaching or ploughing. The soil contamination resulted in the incinerator in question being closed down. Other studies cited suggest that a proportion of dioxins and furans in soils may volatilise or photodegrade.

Comment on data quality:

The data are primarily related to metal and PCDD/PCDF accumulation in soils and primarily relates to specific sites. The Hwang (1990) study does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

Potential effects:

Metals and PCDDs/PCDFs may accumulate in soils due to incinerator emissions. However, incinerators are unlikely to be the primary sources of such contaminants. Other effects may include the loss of soils during site construction.

Water quality

Number of relevant documents: 5

Summary of findings:

Coughanowr et al. (1996) illustrates the preliminary results of a French study assessing the release of pollutants from bottom ash used for roadbed construction. The study indicated that chloride, mercury, pH, total organic carbon, cyanide, fluoride and phenols are released at very low concentrations and are therefore unlikely to have a significant negative impact.

Hwang (1990) provides modelled results for theoretical concentrations of 2,3,7,8-TCDD in either a pond ($4000m^2$ and 5m deep) or large river 800m (300m wide, 5m deep, flowing at 0.5m/s) from the stack emissions of a conceptual incinerator burning 3000 T of waste per day. This is larger than any UK facility, and the release rate of dioxins per tonne of waste processed is approximately twice the

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

current UK value (see section on air quality). The resulting concentrations are as follows:

Pond = 1.6×10^{-6} ng/l River = 1.6×10^{-10} ng/l

In view of typical UK facility size and emission rates, actual levels are likely to be lower than these values. White et al (1995) suggest that water emissions only arise from wet gas scrubbing operations and are usually treated on-site. Volumes may range from 200-770l per tonne of waste input. (Comment: This no longer occurs at UK facilities – see Chapter 4)

Lisk (1988) cites a number of studies suggesting that incinerator waste water streams contain a range of metals, and undissolved solids. Polycyclic aromatic hydrocarbons, dioxins/furans, alcohols, phenols, aldehydes, ketones, esters, amines, amides and hydrocarbons have also been found in incinerator quench and wastewaters. The Environment Agency (2002f) suggests that the incineration process produces a small quantity of contaminated wastewater requiring treatment. This report discussed effluents from gas scrubbers, which are caustic and contain significant concentrations of heavy metals and organic micropollutants treated prior to disposal. However, such effluents no longer arise from incineration in the UK – see Chapter 4) and these pathways are no longer relevant to the UK situation. Liquid effluents may also arise from quench waters. However, these are relatively uncontaminated and may usually be discharged direct to surface water or sewer.

Rainwater

Feng et al (2000) discussed concentrations of 19 metallic elements in rain and snow samples taken from 8 locations up to 15km downwind of Claremont incinerator, which has an input of 200T/day of MSW from New Hampshire and Vermont in the USA. The study used principal component analysis to identify coal-fired fly ash as the most important source of deposited metals. The incinerator contributed less than 20% of the total variance of the elemental concentrations. The table below illustrates the average concentration of metals in collected rainwater and snow samples. The sites were located in all directions around the incinerator. However, the majority of the sites were located downwind for the majority of the monitoring period.

	Appendix 1	Appendix 1 Appendix 2 Appendix 3				Appendix 4					
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

 Table 8.1
 Substance concentrations in precipitation close to a waste incinerator, USA

Metal	Concentration in Rainwater (µg/l)	Concentration in Snow (μg/l)
Sodium	102	151
Potassium	693	38
Magnesium	52	49
Calcium	128	189
Aluminium	2.2	
Manganese	3.8	10.2
Iron	7.8	100
Boron	1.1	1.2
Strontium	0.37	0.74
Barium	0.39	1.5
Zinc	10.3	9.0
Nickel	0.69	1.3
Copper	0.87	0.62
Lead	0.68	1.2
Vanadium	0.19	0.63
Cadmium	0.034	0.026
Chromium	0.026	0.22
Cobalt	0.018	0.075
Arsenic	0.047	1.0

Comment on data quality:

Data are limited and primarily relate to water quality impacts due to atmospheric emissions. The quantitative data that do exist are either modelled (and therefore uses a range of assumptions) or measured and include more significant contributions from other sources. The Hwang (1990) study does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

Potential effects:

Direct releases to water from incinerators are unlikely to have any major effect due to the low volumes involved and the usual practice of on-site treatment (or sewer discharge). However, accidental releases could potentially impact negatively on small, sensitive water courses. Atmospheric emissions are unlikely to make a significant contribution to levels of pollutants in surface waters.

	Appendix 1		Appendix 2 Appendix 3				Appendix 4				
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Water flow

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects on water flows are likely to be limited to the presence of hard surfaces (buildings and hardstanding) causing increased run-off. As such, adequate drainage systems are likely to be required. The impact of any increase in run-off will depend on the volume of the receiving watercourse and the area of the plant.

Air quality

Number of relevant documents: 13

Summary of findings:

The Environment Agency (2002f) listed the potential emissions of greatest concern from incineration as: carbon dioxide, carbon monoxide, NOx, sulphur dioxide, hydrochloric acid, particulates, heavy metals and dioxins/furans.

Lisk (1988) cites a number of studies suggesting that the following have been found in incinerator gaseous emissions: heavy metals, PCDD/F, polycyclic aromatic compounds. polychlorinated biphenyls, chlorobenzenes. chlorophenols. halogenated organic acids, pthalates, aldehydes, ketones, organic acids, alkanes, alkenes, hydrochloric acid, chlorine, sulphur dioxide, NOx, carbon monoxide, hydrogen cyanide, phosphoric acid, hydrogen fluoride, and phosgene. This list remains relevant, although as noted in Chapter 4, emissions from incinerators have substantially reduced since this study was carried out, and some of these substances will not be at significant levels. ODPM/Enviros (2003) (draft) suggests that the principal air emissions associated with incinerators are acid gases, heavy metals, dioxins/dibenzofurans, carbon dioxide and particulates. Emissions may be minimised by control of combustion conditions (temperature, residence times, excess air etc.) and by end-of-pipe controls such as bag filters, scrubbing and activated carbon injection.

Jones (1994), compares emissions of dioxins and furans, benzene, vinyl chloride, carbon monoxide and non-methane VOCs from a theoretical waste to energy plant and landfill in the USA taking a notional 1500 T of waste per day. The older US information is superseded for the UK situation by the information in Chapter 4 of this study.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Bescombes et al (2001) took measurements of particulate polycyclic aromatic hydrocarbons (PAH) in a French urban area in the vicinity of an incinerator at the weekend and Monday morning to control for the effects of traffic. Monitoring occurred during normal incinerator function and maintenance activity (stop and cooling periods). PAH and total particulate carbon were determined at 3 sites - close by, 2km downwind and 1km upwind. In normal operating conditions similar concentrations were observed at all 3 sites. While the furnace was not running PAH concentrations increased at the sampling point close to the incinerator (some 3 times higher than the other sampling points). Pyrene and retene showed the highest increase. This indicates that other sources and the prevailing meteorology are a more significant influence on ambient levels of PAH.

The incinerator was deemed to be a minor contributor to pollution in the area as a whole as the increases were only noted particularly close by and during the furnace stop period. Total PAH concentration ranges and averages at the 3 sites were as follows: 1km upwind = 15.08 ng/m^3 (8.91-33.92) n=7, Incinerator = 27.69 ng/m^3 (6.26-77.39) n=6, 2.3km downwind = 16.42 ng/m^3 (4.33-62.98) n=7. Total carbon concentrations were as follows: 1km upwind = $17.47 \mu \text{g/m}^3$ (8-36.68) n=7, incinerator = $25.49 \mu \text{g/m}^3$ (15.4-57.15) n=6, 2.3km downwind = $26.2 \mu \text{g/m}^3$ (7.92-104.3) n=7. All highest levels were recorded on the Monday indicating a greater contribution from traffic. The study does not describe the sampling locations in reference to localised PAH sources, for example - main roads, which were subsequently identified as the main source of PAH in the area.

Travis and Blaylock (1995) estimated that, in the USA, 100g of dioxins, less than 1% of the total anthropomorphic emissions of dioxins per year, were emitted due to MSW incineration activities. The paper also suggests that waste incineration accounts for 4% of US anthropogenic Hg emissions. A dispersion study in Vermont is introduced which suggested that maximum ground level mercury concentrations would be 0.57ng/m³ whereas background was 2.5ng/m³. Measured mercury levels in soils, water, forage and produce at another site in Minnesota were below detection limits, suggesting incinerators are not a major mercury source. In terms of dioxins, the paper suggests that the incremental air concentration predicted to occur at a point of individual maximum exposure near an incinerator is 11 fg TEQs/m³ (around 12% of mean background levels in US urban air). Another study cited in which measured levels downwind of an incinerator in Ohio were almost identical to background (106 vs 103 fg/m³). Another study suggested that the maximum human dioxin uptake due to incinerator emissions is 130 times less than from background. Other data suggest that incinerators do not increase local dioxin levels significantly above background.

Lohman and Seigneur (2001) undertook a detailed dispersion modelling study of emissions from dioxin point sources, including incinerators. The study predicted that 84-92% of MSW incinerator dioxin emissions are deposited greater than 100km from the source.

Baldasano and Cremades (1995) is a Spanish predictive air impact study of a proposed replacement larger incinerator for the Island of Mallorca (taking 1800T/d). Maximum ground level particle levels were 1.2 and $1.8\mu g/m^3$ (much less than the allowable maximum of $250\mu g/m^3/d$). The new incinerator will cut ground level concentrations of hydrochloric acid from $10.2-3.8\mu g/m^3$. Dioxin levels are also expected to be cut. The reductions are expected despite much greater capacity of the new process.

	Appendix 1			Appendix 2		Appendix 3			Appendix 4			
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification	

Hwang (1990) provides modelled results for theoretical concentrations of 2,3,7,8-TCDD 200 and 800m from a conceptual incinerator burning 3000 T of waste per day. This is larger than any facility in the UK.

The emission criteria were as follows:

Stack Height = 46m Building Height = 42m Number of stacks = 4 Stack diameter = 4.1m Stack gas temperature = 470oK Stack gas velocity = 11.3m/s Emission rate (vapour form) = 1.9×10^{-8} g/s Emission rate (particulate form) = 1.1×10^{-8} g/s

This corresponds to a total emission rate of 900 ng per tonne of waste processed, compared to the value derived in Chapter 4 of approximately 400 ng per tonne.

The resulting concentrations were as follows:

At 200m – 2.1 × 10^{-6} ng/m³ (vapour phase) and 1.2 × 10^{-6} ng/m³ (particulate phase); At 800m – 1 × 10^{-6} ng/m³ (vapour phase) and 6.5 × 10^{-7} ng/m³ (particulate phase).

Air concentrations due to fly ash disposal to land at 30m and 150m from the disposal site are given. However, no details are given as to the assumptions for these concentrations (e.g. the containment that is assumed to be provided).

White et al (1995) suggests that the most significant emissions from mass burn incineration are acid gases (hydrochloric acid, sulphur dioxide and NOx), carbon dioxide, heavy metals (mercury, cadmium and lead), particulates and PCDDs/PCDFs.

Rabl and Spadaro (in Hester et al (2002)) suggests that an incinerator taking 250,000 tonnes of waste per year, emitting at the EC limit concentrations, with a stack 100m high would produce the following maximum incremental ground level concentrations (Cmax in ng/m^3) within a few kilometres of the source:

Arsenic	0.05
Cadmium	0.16
Chromium	0.13
Mercury	0.19
Nickel	0.65
Lead	0.42
Metals (cadmium to thallium)	0.19
Transition Metals (arsenic to vanadium)	1.93
Dioxins	3.9 × 10 ⁻⁷
Particulates	39
Nitrogen Dioxide	773
Sulphur Dioxide	193
Carbon Monoxide	193
Hydrochloric Acid	39
Hydrogen Fluoride	4

It is also suggested that such concentrations are small in comparison to urban background levels.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Patel and Isaac (2002) is a short paper comparing a range of waste management options for a theoretical county including: (1a) Landfill within county, (1b) Landfill outside county, (2) Recycling - 3 bin kerbside collection and MRF/Composting, (3) Large scale incineration, (4) Recycling, anaerobic digestion and small scale incineration. Histograms are used to compare each option against scenario 1b with respect to global warming, acidification, energy, local NOx emissions (transport, particulate and dioxin emissions). Incineration performs better than landfill with regard to NOx emissions. Particle results were much the same across all scenarios. Dioxin results were similar to the particle results but emissions are to different media.

Comment on data quality:

Air quality issues, especially in relation to dioxins/furans have attracted a great deal of research. Further detailed information is given in Chapter 4. The data are either due to modelling (and hence use assumptions) or relate to measurements at specific sites. The Hwang (1990) study does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

Potential effects:

Incinerators emit a wide range of substances to air which could contribute to aspects of air quality such as acid rain and photochemical smog, as well as potentially contributing to adverse health effects. Under the terms of the waste incineration directive and operating authorisations, emissions to air are controlled to set limits. Control measures may include combustion control (temperatures, excess air and residence times) along with scrubbers and filters etc.

Climate

Number of relevant documents: 7

Summary of findings:

Smith et al (2001) suggests that the mass burn of MSW with no energy recovery results in a net greenhouse gas flux of $181 \text{kgCO}_2 \text{ eq/T}$. However, incineration with power production or CHP results in negative greenhouse gas fluxes of -10 and -348 kgCO₂/T. These results take account of waste transport emissions and emissions avoided from power/heat production. Greenhouse gas emission equivalents only taking the incineration process into account are +230 kgCO₂eq/T.

Baldasano et al (1995) suggests that the provision of a new incinerator on Mallorca will improve greenhouse gas emissions. The existing small incinerator emits around $88,000m^3CO_2/d$. With emissions of methane and carbon dioxide from waste landfilling, this comes to $3,853,000m^3CO_2$ eq/d. The provision of a new incinerator is predicted to produce $990,000m^3CO_2/d$ resulting in an overall net reduction in greenhouse gas emissions.

Jones (1994) suggests that waste to energy plants contribute much less to greenhouse emissions than landfills.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Eschenroeder (2001) discussed greenhouse emissions from a theoretical landfill (both with and without gas abstraction) and incinerator taking 1 tonne of waste per day over 30 years (with 70 yr post-closure period), including emissions off-set from power production. The landfill without gas abstraction contributes 115 times more to the greenhouse effect than a similar capacity of incineration. Landfilling with gas abstraction and utilisation contributes 45 times more to the greenhouse effect than incineration. The study included emissions of carbon dioxide, methane, nitrous oxide and chlorofluorocarbons.

Tsilyannis (1999) suggests that landfill with energy recovery emits slightly more greenhouse gases than WTE or separation/RDF/composting. However, this is based on methane being only 7 times more powerful a greenhouse gas than carbon dioxide.

Patel and Isaac (2002) is a short paper comparing a range of waste management options for a theoretical county including: (1a) Landfill within county, (1b) Landfill outside county, (2) Recycling - 3 bin kerbside collection and MRF/Composting, (3) Large scale incineration, (4) Recycling, anaerobic digestion and small scale incineration. Histograms are used to compare each option against scenario 1b with respect to global warming, acidification, energy, local NOx emissions (transport, particulate and dioxin emissions). Global warming results suggest that the incineration option is preferable, in terms of greenhouse gas emissions, than landfill due to the avoidance of releases of methane from landfill.

The Environment Agency (2002f) suggests that around half the carbon dioxide emitted is of fossil origin (440kgCO₂/T). However, this may be off-set by emissions foregone from power production.

Comment on data quality:

Carbon dioxide emissions data are well founded in measurement and calculation. Data relating to life-cycle effects rely on a range of generic assumptions.

Potential effects:

Incinerators with energy recovery are likely to result an overall reduction in the global warming potential, as the carbon emitted is contemporary and the energy produced may off-set emissions from fossil fuel use. Greenhouse contributions are also significantly less than landfills due to the avoidance of methane emissions.

Acid gases

Number of relevant documents: 0

Summary of findings:

No specific information (see air quality section).

Comment on data quality:

No information.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

Acid gases including NOx, sulphur dioxide, hydrochloric acid and hydrogen fluoride are emitted in significant concentrations large scale incinerators. However, tall stack heights are likely to result in low ground level concentrations such that building structures are unlikely to be affected..

Other environmental effects

Other effects associated with incineration include visual impacts, suspected health effects due to atmospheric emissions, ash disposal issues and litter.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.4 Small Scale Incineration of Pre-sorted Wastes with Energy Recovery

Noise

Number of relevant documents: 2

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that noise at small incinerators may be attributed to vehicle manoeuvring, induced draft fans, air cooled condenser units and steam release valves/pipework. Typical limits are 45-55dB(A) (daytime) and 35-45dB(A) (night-time) as for any other industrial facility.

The Environment Agency's (2002) report suggests that the enclosed nature of the operation limits the potential for noise nuisance at RDF plants but that noise sources may include fans, and gas cleaning systems. However, noise is relatively easily controlled via soundproofing and bunds etc.

Other references including: McLanaghan (2002) and White et al (1995), whilst mentioning RDF plant, do not differentiate environmental effects from small and large scale incineration.

Comment on data quality:

Information is limited and qualitative, focusing primarily on sources and controls.

Potential effects:

Like any industrial plant small scale incinerators will emit noise. The impact of any such noise emissions will be dependant on the distance to sensitive receptors, the loudness, frequency and duration of the noise, the time of day and controls in place. However, given appropriate siting and controls noise nuisance is unlikely. Appropriate controls may include bunds, fences, building soundproofing, sensitive siting, regular plant maintenance and provision of smart reversing alarms (i.e. those which limit their outputs to 5dB(A) above background).

Odour/dust

Number of relevant documents: 2

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that, as at any waste management plant, odours may derive from the wastes received, especially if they are allowed to decompose. As such, odour release is only likely if plant failure occurs or storage times are excessive, leading to a build up of wastes within the facility. Dusts may also be emitted during material transfer (loading/unloading) and via vehicle movements. Dusts may also be emitted from ash, which should be stored in covered containers or within the building. Air pollution control reagents, such as lime, may also cause dust if allowed to escape. Both odours and dusts may be controlled via use of internal air for combustion.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

The Environment Agency (2002f) suggests that the enclosed nature of the operation limits the potential for noise nuisance at RDF plants and that any odours which do arise are likely to derive from fugitive emissions or waste handling operations. However, good plant design, waste handling practices and good housekeeping can eliminate odour. Use of internal air for combustion may also assist in odour control.

Other references including: McLanaghan (2002) and White et al (1995) whilst mentioning RDF plant do not differentiate environmental effects from small and large scale incineration.

Comment on data quality:

Information is limited and qualitative, focusing primarily on sources and controls.

Potential effects:

Odour could potentially arise at small scale incineration plants or RDF manufacturing operations dealing with raw waste materials. Odour may be controlled effectively by enclosure of waste reception areas in buildings and using internal air as combustion air along with minimising waste storage periods and good housekeeping.

Dust may be associated with vehicle movements and waste/material handling. Effects may be minimised by preventing and clearing spillages, road sweeping/bowsing and appropriate storage of dusty materials along with appropriate siting. Given effective controls, dust is unlikely to be an issue at small scale incineration plants.

Fauna

Number of relevant documents: 1

Summary of findings:

Hwang (1990) provides modelled results for theoretical concentrations of 2,3,7,8-TCDD in beef and fish (in a pond and large river) 800m from stack emissions from a conceptual small scale incinerator burning 120 T of waste per day. The assumed emission rate of dioxin is approximately 50 times greater than that identified in Chapter 4 for emissions from small-scale incineration (see section on air quality). The results are as follows:

Beef = 1×10^{-5} mg/kg Fish (pond) = 1.3×10^{-6} mg/kg Fish (river) = 1.4×10^{-10} mg/kg

These are theoretical calculations, and actual levels are likely to be less than 2% of these values.

Modelled concentrations in beef and fish due to fly ash disposal to land at 30m and 150m from the disposal site are given. However, no details are given as to the assumptions for these concentrations (e.g. the containment that is assumed to be provided).

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Comment on data quality:

The data produced by Hwang (1990) are modelled values based on a range of assumptions and theoretical emission sources and receptors. The paper is also based on emissions rates some fifty times higher than would be expected in the UK and it is also limited to 2,3,7,8-TCDD. The Hwang study also does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

Potential effects:

Whilst fauna may accumulate dioxins associated with atmospheric emissions, potential effects on fauna are likely to be limited to habitat loss due to construction of the plant.

Flora

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects on flora are likely to be limited to habitat loss during construction as for any other industrial facility of a similar size.

Soil

Number of relevant documents: 1

Summary of findings:

Hwang (1990) provides modelled results for theoretical deposition rates and soil concentrations of 2,3,7,8-TCDD 200 and 800m from a conceptual small scale incinerator burning 120 T of waste per day. The assumed emission rate of dioxin is approximately 50 times greater than that identified in Chapter 4 for emissions from small-scale incineration (see section on air quality). The results are as follows:

Deposition rate at $200m = 0.168\mu g/m^2/yr$ Deposition rate at $800m = 0.028\mu g/m^2/yr$ Soil concentrations at $800m = 2.4 \times 10^{-5} mg/kg$

Soil concentrations due to fly ash disposal to land at 30m and 150m from the disposal site are given. However, no details are given as to the assumptions for these soil concentrations (e.g. the containment that is assumed to be provided).

	Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification	

Comment on data quality:

The data produced by Hwang are modelled values based on a range of assumptions and theoretical emission sources and receptors. The paper is also based on emissions rates some fifty times higher than would be expected in the UK and it is also limited to 2,3,7,8-TCDD. The Hwang study also does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

Potential effects:

Whilst soils may potentially accumulate dioxins from stack emissions potential effects on soils are likely to be limited to removal/damage during construction and the potential for contamination with polluted run-off.

Water quality

Number of relevant documents: 1

Summary of findings:

Hwang (1990) provides modelled results for theoretical concentrations of 2,3,7,8-TCDD in either a pond or large river 800m from the stack emissions of of a conceptual small scale incinerator burning 120 T of waste per day. The assumed emission rate of dioxin is approximately 50 times greater than that identified in Chapter 4 for emissions from small-scale incineration (see section on air quality). The resulting concentrations are as follows:

Pond = $1.3 \times 10^{-7} \mu g/l$ River = $1.4 \times 10^{-11} \mu g/l$

Concentrations in water due to fly ash disposal to land at 30m and 150m from the disposal site are given. However, no details are given as to the assumptions for these concentrations (e.g. the containment that is assumed to be provided).

The Environment Agency (2002f) suggests that a small quantity of contaminated wastewater is produced by the process requiring treatment. Effluents arise from gas scrubber effluent, are caustic and contain significant concentrations of heavy metals and organic micropollutants. Liquid effluents may also arise from quench waters. However, these are relatively uncontaminated and may usually be discharged direct to surface water or sewer.

Other references including: McLanaghan (2002) and White et al (1995), whilst mentioning RDF plant, do not differentiate environmental effects from small and large scale incineration.

Comment on data quality:

Information is primarily qualitative and limited to controls. The data from Hwang are theoretical and limited to 2,3,7,8-TCDD. The paper is also based on emissions rates about fifty times higher than would be expected in the UK. The Hwang study also does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

Significant effects on water quality are unlikely if effective treatment is provided. Accidental spillages could potentially cause limited effects. The nature and extent of such effects will be dependent on the volume and strength of the discharge, and the volume/sensitivity/use of the receiving water resource.

Water flow

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects on water flows are likely to be limited to the presence of hard surfaces (buildings and hardstanding) causing increased run-off. As such, adequate drainage systems are likely to be required. The impact of any increase in run-off will depend on the volume of the receiving watercourse and the area of the plant.

Air Quality

Number of relevant documents: 3

Summary of findings:

ODPM/Enviros (2003) (draft) states that the principal emissions from small scale incinerators are: acid gases, carbon dioxide, heavy metals, particulates and dioxins/furans.

The Environment Agency (2002f) suggests that emissions may include: carbon dioxide (implicated in climate change), carbon monoxide (a respiratory toxin), nitrogen oxides (implicated in photochemical smogs, acid rain, respiratory irritation and plant damage), sulphur dioxide (implicated in acid rain and as a respiratory irritant), hydrochloric acid (implicated in corrosion), particulates (implicated in smutting and respiratory disorders), heavy metals (which are toxic) and dioxins/furans (which are implicated in human health effects).

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Hwang (1990) provides modelled results for theoretical concentrations of 2,3,7,8-TCDD 200 and 800m from a conceptual small scale incinerators burning 120 T of waste per day. The emission criteria were as follows:

Stack Height = 27.4m Building Height = 27.3m Number of stacks = 2 Stack diameter = 1.2m Stack gas temperature = 543oK Stack gas velocity = 12m/s Emission rate (vapour form) = 1.1×10^{-7} g/s Emission rate (particulate form) = 0.7×10^{-7} g/s

This corresponds to a total emission rate of 130 μ g per tonne of waste processed, compared to the value derived in Chapter 4 of approximately 2.4 μ g per tonne.

The resulting concentrations are as follows:

At 200m – 1.8×10^{-13} g/m³ (vapour phase) and 1.1×10^{-13} g/m³ (particulate phase) At 800m – 8.3×10^{-14} g/m³ (vapour phase) and 4.9×10^{-14} g/m³ (particulate phase)

Air concentrations due to fly ash disposal to land at 30m and 150m from the disposal site are given. However, no details are given as to the assumptions for these air concentrations (e.g. the containment that is assumed to be provided).

Other references including: McLanaghan (2002) and White et al (1995), whilst mentioning RDF plant, do not differentiate environmental effects from small and large scale incineration. However, White et al (1995) suggests that RDF plants emit less heavy metals in their exhaust gases.

Comment on data quality:

Most data are limited to emissions and controls. Data from Hwang (1990) are modelled and theoretical. The paper is also based on emissions rates some two orders of magnitude higher than would be expected in the UK. The Hwang study also does not effectively explain the assumptions or calculation methods used or provide reference values against which the results may be assessed, thereby limiting the study's usefulness.

Potential effects:

Air quality impacts from small scale incineration are likely to be similar to those from large scale mass burn plants with similar pollutants emitted. However, the relatively small scale would suggest lesser potential effects on local air quality. However, this will depend on stack heights and the level of emissions control emplaced.

	Appendix 1			Appendix 2		Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Climate

Number of relevant documents: 2

Summary of findings:

The Environment Agency (2002f) suggests that around half the carbon dioxide emitted is of fossil origin (440kgCO₂/T). However, this may be off-set by emissions avoided from power production.

Smith et al (2001) suggests that, if RDF is used as a fuel for power production, a negative greenhouse gas flux of around -340kgCO₂eq/T is achieved.

Comment on data quality:

Information given in Smith relies on a range of assumptions.

Potential effects:

Small scale incineration for power production is likely to result in positive effects on the emission of greenhouse gases given emissions avoided from burning fossil fuels.

Acid gases

Number of relevant documents: 0

Summary of findings:

See section on air quality above.

Comment on data quality:

See section on air quality above.

Potential effects:

Small scale incineration emits acid gases such as sulphur dioxide, nitrogen oxides, hydrochloric acid and hydrogen fluoride. Any effect on building erosion will depend on the emission concentration and height, weather patterns and proximity to sensitive structures.

Other environmental effects

Other environmental effects may include those associated with litter, visual intrusion and solid residues.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.5 Gasification/Pyrolysis with Energy Recovery

Noise

Number of relevant documents: 3

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that whilst the gasification/pyrolysis process itself is unlikely to be noisy, noise may be attributed to: vehicle manoeuvring, waste sorting, ventilation fans, internal screening/sorting operations; steam turbine units and air cooled condenser units. However, if all noisy operations occur within a building, noise is unlikely to become a nuisance. Nevertheless, if sited in quiet locations or close to sensitive receptors noise abatement may be required. Typical noise limits are 45-55dB(A) (daytime) and 35-45dB(A) (night time) similar to other industrial facilities.

The Environment Agency (2002f) suggests that noise is likely to be similar to other thermal processes and that as long as good modern design of the waste reception facilities are adopted, noise is unlikely to represent a problem.

Enviros 2003 report to Norfolk County Council considered 4 different pyrolysis/gasification systems. The Global Olivine Total Resource Management Scheme (GOTRMS) was said to require a 'buffer zone' for noise attenuation and noise insulation in the turbine hall and roof. The Thermoselect process has noise contained by incorporating soundproofing in the walls. The Compact power Avonmouth facility lies 500m from housing and the noise impact is stated to be 'not significant'. The Environmental Statement for the proposed Brightstar process in Derby suggests that 3 properties would be exposed to noise at more than 5dB above background L90. Under the terms of British Standard BS4142 this could potentially result in complaints. As such, further noise attenuation may need to be provided.

Comment on data quality:

Data primarily relate to controls and potential sources and are non-quantitative.

Potential effects:

Like many industrial processes gasification/pyrolysis may result in emission of noise. The impact of any noise emissions will depend on proximity to sensitive receptors, loudness, frequency, period and timing of noisy activities. Night time operations may be especially problematical as background levels will tend to drop. Controls may include building insulation along with sensitive siting, bunds, fences, limiting operational hours and the use of 'smart' reversing alarms (i.e. those that limit their output to 5dB(A) above background) etc.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Odour/dust

Number of relevant documents: 6

Summary of findings:

ODPM/Enviros (2003) (draft) suggest that there is so little experience of such facilities that it is not possible to judge whether nuisance could be caused. Nevertheless, like all waste management operations, due to the nature of the material being handled, there is always the potential for the release of odours. Restricting the storage of waste to a minimum will reduce the likelihood of odour problems. Dust may be generated from vehicle movements and material handling.

The Environment Agency (2002f) suggests that odour/dust is likely to be similar to other thermal processes and that as long as good modern design of the waste reception facilities is adopted, odour/dust is unlikely to represent a problem.

Enviros 2003 report to Norfolk County Council considered four different pyrolysis/gasification systems. No odour or dust issues arise at the Global Olivine Total Resource Management Scheme (GOTRMS). The Compact Power system allows for waste to be contained within lidded bins and a negative pressure is applied to the waste reception area. The air is then used as combustion air, thereby destroying any odour. The Brightstar system also utilises negative pressure in the reception area and autoclaving prior to pyrolysis cuts odour emissions.

McLanaghan (2002) suggests that odours on the plant periphery are likely to be low as odours tend to be contained within the building with internal air used for combustion.

Hertfordshire (undated) also suggests that odour may be controlled by using internal air for combustion purposes.

Comment on data quality:

Information primarily relates to controls.

Potential effects:

With appropriate controls (e.g. use of internal air for combustion purposes) odours are unlikely to represent a significant problem at pyrolysis/gasification plants. Dust may be similarly controlled, or controlled via good housekeeping (sweeping etc).

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Fauna

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects on fauna are likely to be limited to habitat loss as for any other industrial facility of a similar scale.

Flora

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects on flora are likely to be limited to habitat loss during construction as for any other industrial facility of a similar scale.

Soil

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects on soils are likely to be limited to removal/damage during construction and the potential for contamination with polluted run-off as for any other industrial facility of a similar scale.

	Appendix 1			Appendix 2		Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Water quality

Number of relevant documents: 4

Summary of findings:

The Environment Agency (2002f) suggests that liquid residues are associated with boiler blow-down and wet scrubbing systems for flue gas cleaning. Liquid residues may also derive from the reduction of organic matter and can be highly toxic and, therefore, require specialist disposal.

Enviros' 2003 Report to Norfolk County Council suggests that process water at the Thermoselect plant is treated on-site and that the Compact Power System avoids water contamination issues by collecting process water and injecting it back into the pyrolysis chamber, thereby evaporating it.

McLanaghan (2002) suggests that there are no process releases to water but pyrolysis results in water separating from the liquid fuel and requires treatment.

Hertfordshire (undated) states that the Brightstar system has minimal wastewater emissions as it uses very little water and all water is recycled through a water treatment system. Water recovered from the processed waste stream is treated and used as a biofertiliser, for process cooling or for wash down purposes.

Comment on data quality:

Information primarily relates to sources and controls and is qualitative.

Potential effects:

Given the low volumes of water involved and the current practice of on-site treatment, effects on water quality are considered unlikely unless an accidental release occurs.

Water flow

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects on water flows are likely to be limited to the presence of hard surfaces (buildings and hardstanding) causing increased run-off as for any industrial facility of a similar size. As such, adequate drainage systems are likely to be required. The impact of any increase in run-off will depend on the volume of the receiving watercourse and the area of the plant.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Air Quality

Number of relevant documents: 5

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that emissions from gasification/pyrolysis are comparable with other forms of thermal treatment and, indeed, may be lower, although there is not enough evidence as yet to be sure of this. Emissions are likely to include acid gases, carbon dioxide, heavy metals, particulates and dioxins/furans. Air pollution control systems are required.

The Environment Agency (2002f) suggests that gasification/pyrolysis systems can produce less than 10% of the volume of gaseous emissions when compared to mass burn incineration; however, a more detailed evaluation is provided in Chapter 4 of this report. Emissions include: dust, total organic carbon, hydrochloric acid, hydrogen fluoride, sulphur dioxide, NOx, mercury, cadmium, thallium, heavy metals, dioxins, furans and carbon monoxide.

Enviros 2003 report to Norfolk County Council considered four different pyrolysis/gasification systems. Information on the GOTRMS indicated emissions of NOx, hydrochloric acid, hydrogen fluoride, particulates, dioxins/furans, carbon monoxide, volatile organic compounds, and metals and that 90% of SOx could be removed via scrubbing. Low NOx emissions are achieved via selective non-catalytic reduction. A Thermoselect process in Germany was noted to have emitted heavy metals at 10 times the legal limit until filtration was incorporated. Comments on the Brightstar plant in Woolongong, Australia, suggest that the plant emissions did not meet WID standards for NOx, total organic carbon and carbon dioxide.

McLanaghan (2002) comments that gasification/pyrolysis plants are required to operate in accordance with the WID and that gas produced by gasification contains toxic components. However, the gas produced in a gasification system is an intermediate stage in the process. It is burnt to produce energy, with resultant emissions to air as discussed in Chapter 4.

Hertfordshire (undated) suggests that the Brightstar system emits hydrochloric acid, dioxins/furans, NOx, sulphur dioxide, particulates, cadmium, thallium, mercury, lead and hydrogen sulphide at levels which will meet or improve on the current requirements. It also suggests that emissions are substantially lower than for conventional mass burn incineration.

Comment on data quality:

Quantitative data primarily relate to emissions. There are no data on the effects of these emissions on the environment. Available information relates to controls and emissions.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

The gasification/pyrolysis process emits a wide range of atmospheric pollutants including acid gases, volatile organic carbons, carbon oxides, dioxins/furans and metals. The effect of such emissions will depend on the size of the plant, the level of emission control, the level of gas pre-cleaning, the quality of the receiving air, stack heights and weather patterns. Nevertheless, emissions appear to be an improvement on those associated with mass burn systems.

Climate

Number of relevant documents: 2

Summary of findings:

The Environment Agency (2002f) suggests that around half the carbon dioxide emitted from the process is of fossil origin $(440 kg CO_2/T)$ and the other half $(440 kg CO_2/T)$ is contemporary 'short cycle' carbon which has minimal effect on climate change. Carbon dioxide will also be emitted by waste and product transport. However, the fossil contribution is likely to be off-set by energy generation.

This is consistent with the findings of Smith et al (2001) that pyrolysis/gasification results in a very slightly negative greenhouse gas flux of $-3kgCO_2/T$ taking account of emissions foregone from fossil fuel power production along with transport associated emissions.

Comment on data quality:

Smith et al (2001) rely on a number of assumptions which may not necessarily apply to all plant. The Environment Agency's report does not state from where its figures are derived.

Potential effects:

From the above it appears that gasification/pyrolysis is likely to have a broadly neutral effect on greenhouse gas emissions.

Acid gases

Number of relevant documents: see air quality section above.

Summary of findings:

See air quality section above.

Comment on data quality:

See air quality section above.

Potential effects:

Pyrolysis/ gasification emits a number of acid gases including SOx, NOx, hydrochloric acid and hydrogen fluoride. Associated vehicles also emit NOx. The

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

potential for damage to buildings will depend on proximity to sensitive structures, stack heights, emission controls and weather conditions.

Other environmental effects

ODPM/Enviros (2003) (draft) suggests that other effects may include: traffic, litter and visual intrusion.

Other references suggest potential impacts from solid residues and visual effects.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.6 Landfill with Landfill Gas Flaring and/or Energy Recovery

Noise

Number of relevant documents: 2

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that noise problems at landfills tend to derive from vehicle manoeuvring, site preparation works/engineering and landfill gas flares/engines (especially at night). However, landfills are not inherently noisy and most problems arise from poorly maintained plant. Noise limits at receptors are typically 45-55dB(A) (daytime) and 35-45dB(A) (night-time) or 5-10dB(A) above background LA₉₀.

Redfearn et al (2000) suggests that noise complaints amount to around 5% of all complaints received at landfills. All complaints were isolated and tended to relate to motorcycle trespass and reversing alarms. Other noise sources complained about included: bird scarers, operation before working hours, early morning lorry movements, use of old plant, extraction fans, vehicle repairs, refrigerated vehicles at night, plant operative training and gas collection pipe faults. This corresponds to approximately 0.2 noise complaints per site per year.

Comment on data quality:

A limited amount of information on incremental noise levels and noise complaints is available.

Potential effects:

Noise may potentially be an issue at landfill sites depending on the proximity of sensitive receptors, hours of operation and intensity of operations. However, the frequency of complaints suggests that noise is a limited issue. Noise may be controlled by sensitive siting, the use of bunds and fences, plant maintenance, smart reversing alarms (i.e. those which limit their output to 5dB(A) above background) and phasing to allow screening by previous landfill cells.

Odour/dust

Number of relevant documents: 6

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that the primary impacts of landfills regarding odour nuisance occur when landfill gas is allowed to escape from outside the influence of the gas control system. However, the presence of putrescible wastes themselves may also give rise to odour emissions. In some cases landfill odours have been detected over 1km away and over 50% of complaints made to landfills relate to odour. Nevertheless, odour problems may be reduced by good site and landfill gas management practices. Dust may also be generated particularly when waste is unloaded and from vehicle movements, especially over unconsolidated roads, and during material handling operations for site engineering works. However, landfill dusts tend to be relatively coarse, do not disperse widely

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

and landfills are, therefore, generally not associated with dust nuisance. Indeed, only around 1% of complaints to landfills relate to dust.

Redfearn et al (2000) suggests that 59% of all complaints made to 46 landfills (including, active, inactive, inert and putrescible waste sites) related to odour. 20 of the 46 sites had received odour complaints. At most sites odour complaints were isolated but 7 sites caused more than 5 complaints/year and 1 caused 80 complaints in a year (due to disposal of sewage cake). The sites receiving complaints did not appear to be closer to residential areas than those sites which did not receive complaints. Not all complaints could be substantiated. This corresponds to approximately 2.5 odour complaints per site per year (likely to be due to both MSW and other waste streams). This average value conceals a wide variation from site to site, with some sites attracting a much larger number of complaints, and many attracting none. The Environment Agency (2004) suggest that landfill sites generate approximately 10,000 odour complaints per year.

Odour complaints were caused by a range of factors including: disposal of especially odorous wastes (sewage sludge, contaminated soils, wastes stored for long periods), landfill gas escape, poor daily cover, and poor gas abstraction. (NB. Leachate handling and storage may also be odorous). Nearly half of the complaints occurred between the months of November and January due to the relatively high frequency of calm, stable, night-time atmospheric conditions. Only around 1% of the complaints were due to dust. Dusty episodes were associated with high winds during ground preparation works.

Sellwood (2000) and Sellwood/Redfearn (2001) suggest that landfill odours may derive from landfill gas escapes from large areas or point sources such as leachate wells, and from odorous wastes. However, the most important emissions tend to derive from area releases of landfill gas. The primary odorants tend to be in the following groups (in approximate order of strength): organosulphurs, esters, organic acids, hydrocarbons and alcohols. Impacts also tend to be greatest on cool calm winter nights in stable atmospheric conditions. Sellwood/Redfearn (2001) also suggests that risks of odour impact are not generally high beyond 500m from the site. This reference also indicates that the primary sources of dust at landfills are unconsolidated haul roads and vehicle movements, along with engineering works, wind erosion from unvegetated areas or stockpiles and tipping operations. Dust impacts are also only likely to occur in dry summer conditions in strong winds. Dust impacts and allergic reactions at higher levels of dust. Dust soiling impacts are only likely to arise within 250m of the source.

Various papers characterise the odorous constituents of landfill gas. For example, Parker (1983 (ii)) suggests that odorants in landfill gas may include: saturated and unsaturated aliphatic hydrocarbons; esters, alkyl substituted benzenes, terpenes, volatile sulphur compounds and chlorinated hydrocarbons. The esters and sulphides tend to be the most pungent. El-Fadel et al (1997) suggests that odours at landfill sites are predominantly a result of esters, hydrogen sulphide, organosulphurs, alkylbenzenes, limonene and other hydrocarbons present in landfill gas. The odour character of a gas can vary widely with waste composition and age. The extent of odour spread depends primarily on weather conditions.

The Environment Agency (2002e) characterised the odorous constituents of landfill gas based on measurements of gas generated from current UK landfills, highlighting the 25 most odorous compounds typically found in landfill gas.

	Appendix 1 Appendix 2					Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Comment on data quality:

Quantitative data focus primarily on emission rates or landfill gas constituents. Information on odour complaints is also available.

Potential effects:

Odour is one of the primary impacts of landfilling operations and, in some circumstances, can result in complaint from distances of up to 500m away. Any effect, however, will depend on the nature of the wastes being deposited, the landfill design, the degree of landfill gas collection, weather conditions and the proximity/orientation of sensitive receptors. As collection and combustion of landfill gas becomes more widespread throughout the landfill industry, odour impacts from landfill may be expected to decrease.

Fauna

Number of relevant documents: 4

Summary of findings:

Brown et al (undated) describes a study of a leachate release to a small intermittent stream from a landfill site in the US. 100 Brook Trout fingerlings were exposed to a range of dilutions of leachate from 1% to 100% for 96hrs after acclimatisation. No fish died, no 'coughing' was observed (where fish attempt to clear their gills). Chemical data are also presented. It was concluded that this particular leachate was relatively benign despite some parameters being over USEPA criteria for freshwater aquatic life (including total Kjeldahl nitrogen, mercury, iron, manganese and sodium).

Rutherford et al (2000) studied treated leachate discharged into a stream at Highway 101 Landfill on Nova Scotia (taking 700 T of waste per day and discharging138m³ of leachate each day). Historic studies in 1989 suggested that the leachate (then treated in lagoons and discharged via a wetland into a river) was highly toxic to Rainbow and Brook Trout, and moderately toxic to Daphnia magna. This toxicity was mainly due to high ammonia levels which had dropped from 21.7mg/l in the leachate in 1989 to 2.8-4.1mg/l in 1993 due to treatment. Subsequent to this study, further new leachate treatment systems were installed and toxicity showed a marked improvement. Samples were not acutely toxic to Trout and had no effect on survival and reproduction of *Ceriodaphnia dubia*. The discharge actually had a buffering effect on the toxicity of upstream waters affected by high pH levels. However, the benthic macroinvertebrate community was affected by the discharge, as shown by an increase in the number of midge larvae due to increased organic loads. This effect was localised to the 50m stretch below the discharge. Aluminium concentrations may also have contributed to the effect.

ODPM/Enviros (2002) (draft) suggests that landfills may become sources of flies, vermin (rodents and foxes) and birds due to the presence of food materials for such species. Fly infestations, which can migrate off-site to surrounding areas, may occur in hot summer weather conditions but can also originate from further up the waste stream in the event of long storage periods prior to disposal. Rodents are generally not a problem where compaction and effective daily cover are used. However, larger animals, like foxes, occasionally use landfills as foraging territory. Birds (primarily gulls and crows) are attracted to landfills for food. Birds may be associated with noise nuisance and soiling of property. There may also be effects associated with loss of habitats, pollution of watercourses and human disturbance.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Positive effects may occur following restoration during which habitats may be created which would be attractive to a wide range of species (specific examples include adders, badgers, various birds, butterflies and moths).

Redfearn et al (2000) suggest that flies account for around 24% of complaints made to 46 landfill sites surveyed. Most complaints were isolated. However, two sites received more than five complaints per year with one receiving around 75/yr. Many fly complaints could not be positively linked to the landfills concerned. Most complaints were made in the summer (90% from May to September). Very few complaints (<1%) were made regarding rodents and birds). The bird complaints were all made in winter when gulls tend to congregate inland.

Comment on data quality:

Quantitative data primarily relate to individual sites and circumstances and cannot, therefore, be reliably related to any other sites or operations. Qualitative information mostly deals with nuisance species.

Potential effects:

From the above studies landfills may potentially have a negative effect on fauna including: toxicity of leachate to aquatic organisms (in the event of a direct release), loss of habitat and encouraging vermin/birds/flies. Such effects may be minimised by sensitive siting, vermin control and leachate control. In the longer term, landfills may provide new habitats for a wide range of animals upon restoration.

Flora

Number of relevant documents: 4

Summary of findings:

The main potential effect of landfills on flora arises from migration of landfill gas. This can result in restrictions in root growth due to the reduction of oxygen, and the presence of carbon dioxide and trace gases (in particular, ethylene) (Holley and Phillips, 1996). Neumann and Christensen provide a review of issues relating to the potential effects of landfill gas on vegetation (Neumann and Christensen, 1996). They highlight the main issue as being asphyxiation resulting from reduction of soil oxygen levels, with excessive carbon dioxide levels also potentially being harmful to plants directly or indirectly, as a result of pH change. Effects are reported over a distance of up to 250 metres from a landfill. Neumann and Christensen indicate that significant damage to vegetation can be prevented by proper design of gas control systems, completion of the landfill, and selection of tolerant plant species on and near the landfill itself.

Arthur et al (1985) described a study in which tomato plant roots were exposed to simulated landfill gas mixtures previously measured in landfill soil covers and associated with poor plant growth. A concentration of 18% carbon dioxide or greater, exceeded in 30% of the 32 landfills examined at that time, caused reduced growth and visible symptoms after 1 week regardless of oxygen levels (indicating a toxic response). 32% carbon dioxide resulted in plant death. Methane, at 20% or above, found in >25% of the 32 landfills examined, whilst not directly toxic, was associated with oxygen depletion and plant decline.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

These effects could only be significant in the near vicinity of a landfill site, in the event of migration of landfill gas. El-Fadel et al (1997) suggest that sub-surface migration of landfill gas causing vegetation dieback is well documented. The damage is caused primarily by displacement of soil oxygen by landfill gas resulting in asphyxia. Methane oxidation by bacteria in soils may also contribute to plant damage by using oxygen, increasing soil temperatures and producing carbon dioxide. Trace toxic components in landfill gas may also have an effect. The nature of the restoration soil (thickness, composition, compaction and moisture content) will also affect plant growth.

Shrive et al (1994) is a Canadian study which considered photosynthesis, stomatal conductance and stem growth of Red Maple and Hybrid Poplar irrigated with rain or leachate (with a low metal content and high ionic strength). The study took place over 2 consecutive growing seasons. Three variable factors were used including: irrigant type (leachate or rain), application mode (spray, surface, subsurface) and application rate. Photosynthesis and growth rates for Red Maple did not change significantly with irrigant type. For Hybrid Poplar, photosynthesis rates were unaffected but stem growth did increase significantly with leachate irrigation. Direct spraying of leaves with leachate containing phytotoxic compounds did not induce phytotoxic symptoms.

ODPM/Enviros 2003 (draft) suggests that effects on flora may be associated with loss of habitats, dust deposition, or pollution of watercourses. Vegetation may also be compromised if sub-surface landfill gas migration is allowed to occur. Positive effects may occur following restoration during which habitats may be created attracting various flowering plants and grasses.

Comment on data quality:

Quantitative data relate to individual sites and circumstances and cannot, therefore, be reliably related to any other sites or operations.

Potential effects:

From the above studies landfills may potentially have a number of negative effects on flora, primarily related to the risk of landfill gas migration. Direct contact with untreated leachate is unlikely to occur away from the landfill site itself. Loss of habitat could also be significant, although many landfills are sited to re-use industrial sites, making this a less significant issue. Such effects may be minimised by sensitive siting away from sensitive habitats, landfill gas management and leachate control. In the long term, landfills may provide new habitats for a wide range of plants upon restoration.

Soils

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

Despite a lack of literature evidence, soils may be significantly impacted by landfilling operations, particularly due to soil removal and storage during site construction and engineering, due to the land areas and soil volumes potentially affected. Effects on soils may also be caused through leachate contamination. Such effects may be reduced by ensuring appropriate soil handling and storage techniques, the avoidance of compaction and the prevention of leachate spillages.

Water quality

Number of relevant documents: 9

Summary of findings:

Comprehensive information on the constituents of landfill leachate is provided in Robinson (2003). These are summarised by ODPM/Enviros (2003) (draft) as organic compounds, ammonia, nutrients, heavy metals, chloride, and suspended solids. However, any effect will depend on the volume and strength of the discharge as well as the volume/quality/use of the receiving water. Local water quality may also be affected by surface run-off from landfills in terms of suspended solids and litter.

A landmark review of the effect of landfill on groundwater quality was carried out in 1978 (Department of the Environment, 1978). This report concluded that:

- "(a) pollution plumes around landfill sites are quite often restricted in extent;
- (b) the site geology and hydrogeology, especially the presence of an unsaturated zone, are of great significance in determining the degree of attenuation of leachates;
- (c) attenuation mechanisms (defined broadly to include dilution) are available in the landfill and underlying strate, which are extremely beneficial if used with discretion

Borehole and surface water quality monitoring is carried out at landfills to confirm whether adverse impacts on water quality are occurring. Robinson et al. (2000) reported on a 16 year programme of monitoring at a landfill site in Kent located close to a sensitive river. This study identified trace levels of tritium in the nearby river, suggesting that leachate was reaching the river. However, no measurable adverse effect on river water quality could be detected. A minor effect on groundwater quality was detected at one location.

Riediker et al (2000) is a Swiss study investigating leachate and groundwater samples collected from the pollution plumes of four Swiss landfills in order to characterise an additional pathway for benzenesulphonates and naphthalenesulfonates (BS and NS) into the environment. Results showed that landfill sites are point sources for BS and NS in the aquatic environment. BS and NS were measured in the leachate at a wide concentration range from a few micrograms per litre up to several milligrams per litre, depending on the composition of the deposited material. Their contribution to the leachate dissolved organic carbon was from below 1% to around 30%. BS and NS were also found in

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

groundwater samples contaminated by percolating leachates. The concentrations ranged from 30ng/l to $43\mu g/l$, and contributed less than 1% to the groundwater dissolved organic carbon. Of the four landfills sampled, one was loam lined, two were lined with a layer of bituminous concrete and one was unlined. However, the levels found indicate that BS and NS are negligible groundwater contaminants given their low ecotoxicological risk. Aerobic groundwaters allow both BS and NS to be broken down.

White et al (1995) suggest that around 13% of the rainfall falling on a site will emerge as leachate. For sites in Germany receiving an average rainfall of 750mm/yr this would result in around 100l of leachate produced per m^2 of landfill surface. Further estimates assuming a 20m waste depth and density of waste at $1T/m^3$ would result in leachate production of 5 litres per tonne per year, or 150 litres per tonne over a 30 year leachate producing period. This is consistent with the estimate of 110 litres per tonne given in Chapter 4. Leachates typically contain elevated levels of organic materials, toxic trace organics and heavy metals but compositions vary with waste type and age.

Other literature identified relates to studies on unlined landfill sites, which are not representative of current UK landfill practices. US practice, as considered in the Borden (1989) study, is different from UK practice, and UK practice has changed dramatically over the past 30 years. Current landfills are restricted in their location to those insensitive to groundwater; designed via probabilistic risk assessment such that their impact on groundwater is acceptable; and regulated to operate within the designed parameters. Studies on landfills not designed or operated to current UK standards are set out below.

Mikac et al (1998) describes a study on a Croatian landfill outside Zagreb containing 5Mt of, primarily, MSW (with some industrial waste). The landfill has no liner, lies on very permeable alluvial sediments (2-4m above the saturated zone) and is located a 2-3km away from a protected groundwater zone. Monitoring using 20 piezometers around the landfill was carried out for - conductivity, pH, oxygen, ammonia, nitrate, nitrite, chloride, sulphate, chemical oxygen demand (COD), dissolved organic carbon (DOC), total oils, sodium, calcium, magnesium, potassium, iron, manganese, cadmium, lead, copper, zinc and mercury. The monitoring suggested that an anaerobic plume stretched up to 1200m from the landfill in the direction of groundwater flow, to a depth of up to 60m, and that the aquifer was contaminated particularly with regards ammonia, organic matter, iron, manganese and, in some areas, cadmium. Most metals were significantly attenuated in the soil below the landfill. Ranges of selected determinands showing increased levels in the groundwater are presented below for the leachate, the contaminated groundwater and uncontaminated groundwater in mg/l except where specified.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Parameter	Leachate	Contaminated Groundwater	Uncontaminated Groundwater
Ammonia	109-370	0-48	0.12-0.25
Chloride	390-2480	10-135	3-63
COD	1500-6210	0.9-176	5-25
DOC	163-449	0.4-56	0.1-1
Total Oils	2-7.4	0.02-0.3	-
Sodium	-	4-70	<5-21
Calcium	-	72-211	<5-138
Magnesium	-	5-53	<2-21
Potassium	-	0.7-44	0.6-1.2
Iron	-	<0.03-120	0.5-25
Manganese	-	<0.01-3.5	0.02-1.3
Cadmium (µg/l)	0.4-48	0.01-0.17	<2-0.6

In terms of anaerobic zones outside the landfill an iron reducing zone spread to a distance of around 200m, an iron/manganese reducing zone spread to a distance of around 600m and a nitrate/manganese reducing zone spread beyond 1200m down hydraulic gradient from the site.

EI-Fadel (1997) suggests that leachate can be a significant threat to groundwater in the absence of a collection system. It is speculated that municipal landfills in the US could become a bigger groundwater contamination problem than hazardous waste landfills. Landfill gas may also contaminate groundwater with soluble carbon dioxide. Trace landfill gas compounds, such as vinyl chloride, have been found in groundwater at distance from landfill sites.

Borden et al (1989) carried out a study of 71 municipal sanitary landfills in North Carolina, USA, looking at ground water and surface water contamination. The main findings were that leachate compositions are highly variable depending on site specific conditions including waste composition, pH, temperature, nutrients and degree of decomposition. Violations of water quality standards for both inorganic and organic pollutants were found at 53% of the sites with adequate monitoring data. However, the severity of pollution was highly variable with most sites requiring a 50% reduction or less in pollution concentration to prevent violation. However, a few landfills exhibited extremely high levels of contamination that could exceed the relevant standard by 10,000 times or more. Nevertheless, the contamination within both ground and surface water bodies affected by landfill contamination were orders of magnitude lower than concentrations reported for leachate indicating significant pollution attenuation/dilution outside the landfill.

In terms of surface water results many inorganic substances showed substantially higher values downstream from landfills than upstream. On average zinc concentrations rose from 0.053mg/l to 0.341mg/l. Significant increases were also reported for manganese (0.3-2mg/l), turbidity (42-210 Jackson Turbidity Units (JTU)), and iron (3-12.5mg/l). Other noted increases were for chloride (13-44mg/l), total organic carbon (11-32.5mg/l), conductivity (141-414mmhos), alkalinity (39-97mg/l), total dissolved solids (120-285mg/l) and fluoride (0.1-0.16mg/l). Slight

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

increases in lead (0.03-0.033mg/l), arsenic (0.05-0.052mg/l) and chromium (0.012-0.016mg/l) were also reported. However, the majority of measurements were below detection limits for heavy metals.

Average groundwater quality data showed similar trends using surface water upstream as a benchmark. Zinc increased by 2600% (0.053-1.44mg/l) and major increases were noted for conductivity (141-295mmhos), total dissolved solids (120-204mg/l), total organic carbon (11-32mg/l), barium (0.13- 0.2mg/l), cadmium (0.005-0.0056mg/l), chromium (0.012-0.02mg/l) and lead (0.03-0.078mg/l). However, the high iron, manganese, zinc, cadmium, chromium and lead in both surface and groundwaters could be due to geochemical processes. Iron, manganese and pH regularly violated groundwater standards. However, for iron and manganese this may have been due to collection of clay particles. Lead and zinc had a 90% probability of exceeding standards close to landfills. Around 30% of the landfills examined had heavy metal concentrations (excluding iron and manganese) exceeding water quality standards at 1 or more wells.

In terms of organic pollution 36 landfills were monitored and 14 (39%) reported organic priority pollutants within the groundwater. The most commonly reported organics in groundwater samples included: chlorinated solvents (3 sites), petroleum hydrocarbons (3 sites) and pesticides (2 sites). Two sites also showed high levels of fatty acids from anaerobic waste degradation which could render water supplies unfit for human consumption due to odour. The paper states that landfills pose a significant threat to ground water quality due to organic contamination. However, the data are skewed by high concentrations at very few sites whereas the majority of sites exhibit no contamination.

The Borden paper also reviews some other studies in relation to groundwater contamination from landfills. One study suggested that, at one site, biotransformation and dilution were significant in reducing contaminants in the groundwater. Chloride was cut by 75% between 10 and 200m from a landfill while biological and chemical oxygen demands reduced by 99%. Iron and manganese were also attenuated. Another modelling study suggested that most organics would be removed within a few meters of a landfill perimeter within a sand aquifer. Field studies confirmed this, suggesting drops in chemical oxygen demand from 2000mg/l at the edge of the fill to 150-175mg/l at a distance of 2m. Further reduction at greater distances was also observed due to dilution. Another study suggested a decrease in leachate concentrations in both time and distance from the landfill. Total organic carbon and hydrocarbon concentrations reduced significantly. However, detectable levels of benzene were still present 950m from the site $(6.9\mug/l near the landfill dropping to 0.4\mug/l 950m away)$.

Comment on data quality:

Despite a relatively high number of quantitative studies much of the research has been carried out on sites which are not representative of current UK practice, and the results cannot be extrapolated to other operations. Where studies have aggregated average data (i.e. Borden et al 1989), the containment measures incorporated at the landfill sites are not specified, distances to downstream monitoring points are not indicated and comparison of groundwater results to upstream surface water quality is not valid.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

The release of leachate or contaminated surface drainage waters to surface or groundwaters could potentially have significant effects on water quality. However, the degree of any effect will depend on the volume and strength of the release, attenuation of any releases, and the volume/quality/use of the receiving waters. Modern landfills are now required to contain and collect leachate by using a lining, capping and drainage system. Leachates from MSW landfills also require treatment either on-site or via discharge to sewer. Surface drainage should also be managed to keep clean and dirty waters separate and treat dirty waters as leachate.

Water flow

Number of relevant documents: 1

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that landfills may affect water flows by disrupting surface topography, excavations occurring below the water table and via dewatering operations.

Comment on data quality:

Information is limited and qualitative.

Potential effects:

Landfills may affect water flows by disrupting surface topography, excavations occurring below the water table and via dewatering operations. The degree of any such effects will depend on the area of the landfill, the depth of excavation/filling, the presence of a cap, the proximity to and volume/sensitivity of local surface and groundwater resources and the degree of topographical alteration. Such effects may be minimised by sensitive site design and provision of adequate drainage infrastructure

Air Quality

Number of relevant documents: 7

Summary of findings:

Patel and Isaac (2002) is a short paper comparing a range of waste management options for a theoretical county including: (1a) Landfill within county, (1b) Landfill outside county, (2) Recycling - 3 bin kerbside collection and MRF/Composting, (3) Large scale incineration, (4) Recycling, anaerobic digestion and small scale incineration. Histograms are used to compare each option against scenario 1b with respect to global warming, acidification, energy, local NOx emissions (transport, particulate and dioxin emissions). NOx results suggest that landfilling is worse than incineration or recycling/digestion/incineration. Particle results were much the same across all scenarios but sources vary. Dioxin results were similar to the particle results but emissions are to different media.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

ODPM/Enviros (2003) (draft) suggests that the primary emissions of concern to air quality from landfills include toxic trace constituents in landfill gas (which may potentially be linked to health effects) and combustion products from landfill gas flaring/ utilisation (including: SOx, NOx, COx, unburned hydrocarbons, nonmethane volatile organic compounds, acid gases, particulates and dioxins/furans). However, of these sulphur dioxide, sulphur trioxide, hydrogen fluoride, hydrochloric acid, particulates and dioxins/furans tend to be released in concentrations that are insignificant in terms of effects. Emissions of NOx, carbon monoxide and nonmethane volatile organic compounds / unburned hydrocarbons can be significant especially if combustion is inefficient. In general, flares would be expected to emit less pollutants than gas engines due to more efficient combustion. Gas engines, megawatt for megawatt tend to emit similar levels of pollutants as coal or oil generators. However, any emissions may be off-set by those foregone from burning fossil fuels for power/energy purposes. Nevertheless, the plant must be designed and sited so as to prevent local air quality deteriorating.

A comprehensive study of the trace components of landfill gas was recently completed by the Environment Agency (2002e). This study prioritised the trace constituents into those of greatest concern with regard to potential health effects, and those with the greatest potential for giving rise to odours. This information largely supersedes earlier compilations such as that of White et al. (1995), Parker (1983(ii)) and others. A similar database has been compiled by Redfearn and Roberts (2002).

White et al (1995) suggest that landfill gas contains a range of trace constituents at varying concentrations, the most important of which are hydrogen sulphide, vinyl chloride, benzene, toluene, trichloroethane and mercaptans (this information is updated by more recent Environment Agency research). A typical gas collection efficiency is stated as 40% (although reported ranges are from 20-90%). As such, it is estimated that around 60% (or 10-80%) of the landfill gas within a landfill site will be released. Emissions from gas combustion may include particulates, carbon monoxide, unburned hydrocarbons, NOx, hydrochloric acid, sulphur dioxide, dioxins and furans.

Parker (1983 (ii)) suggests that landfill gas is made up of methane, carbon dioxide, nitrogen, oxygen, hydrocarbons, hydrogen, hydrogen sulphide, carbon monoxide and other trace compounds.

El-Fadel (1997) suggests that a wide range of trace organic gases are released from volatilisation of existing chemicals within the waste or from microbial biodegradation products. Such emissions of volatile organic carbons may potentially increase cancer risks and contribute to ambient ozone formation.

Jones (1994) comparing emissions from a theoretical WTE plant and landfill in the USA taking 1500 T of waste pa suggests that landfill combustion plant releases less toxic equivalents of dioxins/furans than WTE plant by a factor of between around 10 and 20. However, benzene and vinyl chloride emissions are much higher as are emissions of carbon monoxide and non-methane volatile organic compounds. NOx releases are, however, generally slightly lower for the landfill. However, emissions could contribute to the generation of ground level ozone.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Tsilyannis(1999) suggests that in comparison to WTE, and mechanical separation with RDF incineration and composting landfilling with energy recovery produces only slightly higher air emissions. Landfilling is considered not to release heavy metals to atmosphere but does release dioxins and furans at around the same rate as the other two options. PCBs are also emitted along with more than 100 volatile compounds, the primary ones of concern being: hydrogen sulphide, vinyl chloride, benzene, toluene, tetrachloroethene, xylenes, and mercaptans. These have been found on-site at levels above occupational exposure limits.

Further information on landfill emissions to air are set out in Chapter 4.

Comment on data quality:

Information on air quality is primarily focused on quantifying gas constituents rather than environmental effects.

Potential effects:

Landfilling may result in a number of effects on local air quality either from surface emissions of volatile organic compounds etc. or from combustion emissions from burning landfill gas. Volatile organic compounds from surface emissions may be associated with odours and, potentially, with health effects. Emissions from landfill gas combustion may, in particular, increase levels of NOx locally, especially if gas is being combusted in engines and exhaust heights are low. Surface emissions may be controlled by gas abstraction, capping and the use of cover materials. Flare/engine emissions may be controlled by efficient combustion conditions, high temperatures and adequate residence times within the combustion zone. Combustion emissions may also be associated with waste vehicles and on-site plant.

Climate

Number of relevant documents: 8

Summary of findings:

A recent report considered the likely future trends in landfill emissions of methane from the UK (DEFRA, 2003). This report estimated the current national emissions of methane from landfills in the UK to be approximately 550,000 tonnes per year, about a quarter of the total UK methane budget. Emissions of carbon dioxide and methane from landfill account for approximately 10% of UK greenhouse gas emissions.

Eschenroeder (2001) discusses atmospheric response to greenhouse emissions from a theoretical US landfill (with and without gas abstraction) and incinerator taking 1 T of waste per day over 30 years (with 70 yr post-closure period). The study includes emissions off-set from power production. Landfill without gas abstraction contributes 115 times more to the greenhouse effect than incineration. Landfill with gas abstraction/utilisation contributes 45 times more. Gases accounted for include: carbon dioxide, methane, nitrous oxide and chlorofluorocarbons.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Patel and Isaac (2002) compared a range of waste management options for a theoretical UK county. Global warming results suggest that the landfill options are result in greater greenhouse gas emissions than all the other options due to the release of methane.

ODPM/Enviros (2003) (draft) suggests that, as landfill gas contains around 60% methane, which is approximately 25 times as potent a greenhouse gas as carbon dioxide (which makes up the rest of the bulk component of landfill gas), landfills can be significant contributors to climate change.

Smith et al (2001) suggest that landfilling of raw untreated MSW results in a net greenhouse gas flux of +330kgCO₂eq/tonne MSW, assuming European average landfill gas collection rates, waste transport and displaced emissions from landfill gas utilisation. However, this depends on the amount of carbon that may break down in the landfill resulting in a range of fluxes from around 40-650kgCO₂eq/T. When carbon sequestration and best practice gas abstraction/restoration layer provision are taken into account landfills may result in overall greenhouse gas fluxes between +72 and +250kgCO₂eq/T.

EI-Fadel (1997) suggests that methane and carbon dioxide fluxes from landfill surfaces may reach 630 and 950kg/m²/yr respectively. Landfills are likely to contribute significantly to global warming due to the emission of methane which is 20-25 times as potent a greenhouse gas as carbon dioxide. It is estimated that methane contributes around 18% of the UK global warming budget. The total UK emissions are estimated as 500million tons methane/yr, of which 40-75 million tons are attributed to emissions from landfills.

Jones (1997) suggests that landfills emit significantly higher amounts of greenhouse gases than WTE plants.

Tsilyannis (1999) suggests that landfill with energy recovery emits slightly more greenhouse gases than WTE or separation/RDF/composting. However, this is based on methane being only 7 times more powerful a greenhouse gas than carbon dioxide, whereas the figure normally applied is 20-25 times, for a 100 year horizon.

Comment on data quality:

Quantitative data tend to rely on a wide range of assumptions but are broadly consistent.

Potential effects:

Landfills make a significant contribution to greenhouse emissions due mainly to emissions of methane. Gas collection and flaring or use for power production helps to reduce this effect, with the result that the overall contribution is approximately 10% of UK greenhouse gas emissions.
	Appendix 1			Appendix 2			Appendix 3			Appendix 4 Anaerobic Digestion ransportation Energy Acidfic	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Acid gases

Number of relevant documents: 1

Summary of findings:

Patel and Isaac (2002) compared a range of waste management options for a theoretical county. Acid gas results suggested that landfill scenarios were significantly worse in terms of acid emissions than incineration or recycling/digestion/incineration. However, landfill performed better than recycling and composting due to power production from landfill gas offsetting emissions from fossil fuel power production.

Comment on data quality:

Specific information is limited and comparative.

Potential effects:

Landfills may release acid gases (especially NOx) from gas flares and engines (see Air Quality section above), as well as that associated waste transport vehicles. Emissions are unlikely to result in building erosion.

Other environmental effects

Other environmental effects associated with landfills may include gas migration/explosion/asphyxiation risks, visual intrusion and litter.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.7 Composting

Noise

Number of relevant documents: 6

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that noise from composting plants is associated with vehicle movements, mechanical turning operations, waste shredding and compost screening. Such operations are inherently noisy and control measures should be applied.

The Environment Agency's 2002 review of waste pre-treatment suggests that there are two main sources of noise at composting operations – shredding and reversing signals for loading shovels. However, windrows themselves can act as effective noise screens and 'smart' reversing alarms (i.e. those that limit themselves to an output of 5dB(A) above background) may be selected.

McLanaghan (2002) suggests that noise levels at composting facilities are generally 'low to medium' i.e. like a farming operation, but can increase to 'high' at times (e.g. during shredding).

The Environment Agency's 2001 Report (P428) on 'Monitoring the Environmental Impacts of Waste Composting Plants' suggests that noise levels may attract complaints from people who live or work within 300m of a composting operation. Noise levels monitored between 5m and 150m from plant are summarised as follows for the 3 sites studied:

Site	Background Level dBL ₉₀ *	Operation	Rating Level dBL _{ar,T} **
Dogsthorpe	42-56	Shredding	66-79
		Turning	66-80
		Screening	64-84
Netley	50-57	Turning	62-81
		Screening	64-96
Morpeth	50-84	Shredding	66-87
		Screening	71-83

Key: L90 = the noise level exceeded for 90% of the time and referred to as the 'background' level.

Lar,T = the rating level, i.e. the noise level due to emission source in question over a specified time period (inclusive of a +5dB adjustment to take account of particularly annoying components (e.g. whines, hisses and distinct impulses).

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Noise levels due to operations ranged between 7 and 46dB(A) above background. As a difference of +10dB(A) is likely to result in complaints, the above noise levels could potentially be associated with nuisance up to 200m from shredding and 300m from screening operations. Nevertheless, windrows may be used to act as effective screens and siting plants 150m+ from sensitive residential areas may prevent nuisance. Carrying out shredding and screening operations within buildings will also help.

Wheeler (2001) looking at 3 compost sites (2 open green waste sites and 1 enclosed mixed waste site) suggests that noise levels from plant used on composting sites are sufficiently high to cause nuisance to neighbours. However, windrows can be used as screens to mitigate noise. This generally results in noise complaints being rare in relation to composting sites. Calculations derived from onsite measured data suggested that unattenuated 1hr noise levels due to composting operations at the enclosed mixed waste facility, at Lynbottom, would reach 'background' levels (assumed to be 50dB – day and 40dB night) within 280m and 119m respectively.

Wheeler and Bourne (2000) suggest that noise levels measured at 3 sites (Dogsthorpe, Morpeth and Netley) by Casella were at least 10dB(A) above background up to 150m away from operations. Given this differential, noise complaints could be expected. Shredding was the noisiest operation with turning/screening being around 6dB quieter. Reversing signals were also a factor and enclosure of shredding within a building at Morpeth significantly reduced emissions.

Comment on data quality:

The quantitative data that exist suffers from a lack of traceability to specific distances from plant. Many papers do not indicate the type of operation or the wastes being processed. However, the data give a good indication of the general distances from composting plants that unattenuated noise may become a problem (i.e. around 300m away).

Potential effects:

Noise is a potential problem, particularly at open composting sites. Particularly loud noise emissions are associated with shredding, turning and screening operations such that nuisance levels may be experienced in the order of 300m from any site if no mitigation is put in place. However, windrows may effectively screen noise emissions and especially noisy operations can be enclosed within buildings.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Odour/dust

Number of relevant documents: 6

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that odour at composting sites has led to the most public complaints about such operations and has resulted in site closures. Odour emissions derive from the delivery of feedstock (especially if it has been stored for long periods), shredding, exhaust air from enclosed systems, the development of anaerobic conditions, dirty areas and roads and untreated pools of leachate. The greatest potential for odour is when materials are not aerated sufficiently, or become too wet, leading to anaerosis and the release of odorants when the material is disturbed. This can occur when facilities are poorly managed or plant failure occurs. At well run facilities odours are reduced by minimising storage periods, maintaining aeration, leachate control and good housekeeping. However, odours at composting sites can never be eliminated. Dust may also be liberated from composting facilities especially when materials are allowed to become too dry and may become a nuisance. Dry materials will liberate dust when being turned, shredded or screened. Vehicle movements may also liberate dust.

The Environment Agency (2002f) suggests that composting operations may result in odour complaints but that this may be minimised through good management. However, for open systems it is not possible to eliminate odours. Odour emissions from enclosed systems may be controlled via biofilters or scrubbers.

McLanaghan (2002) suggests that odour levels are normally 'low to medium' and biofilters may assist in odour control.

The Environment Agency's 2001 report suggests that inhalable dust from composting sites does not represent a risk to either plant operators or to the general public and found no obvious trends between upwind and downwind samples. Odour levels, however, may result in nuisance up to 80m away. Measured levels ranged from 35-6800U/m³ between 1 and 70m from various operations. (NB. The text also cites another study by Fischer where levels varied between 20 and 80,0000U/m³). Grassy/pine odours were the most commonly reported odour characters (most likely from the composting of green wastes). It should be noted, however, that the second highest odour concentration was measured upwind of one of the sites.

Wheeler et al (2001) suggests that inhalable dusts on a range of composting sites were generally below occupational exposure limits and, taking a notional 250µg/m³ limit for off site levels, this level would be reached within 250m. The greatest distance from the Lynbottom, mixed waste composting plant, required to attenuate inhalable dust levels to below this threshold was 199m due to emissions from unloading operations. Deposited dust at Lynbottoms' site boundary averaged $227 \text{mg/m}^2/\text{day}$ (range = 67-499 mg/m²/d). In a normal rural situation deposition rates of $40-60 \text{ mg/m}^2/\text{d}$ would be expected. The rates measured at Lynbottom are within the range expected for an area with heavy industrial/construction activities. Dust deposition tends to increase in the summer. The study also suggested that odour from composting facilities in general is a common cause of complaint and that composting of mixed wastes and source separated organics can be especially problematical. Odour from the Lynbottom enclosed mixed waste plant was estimated to require a distance of 940m to be reduced to $5OU/m^3$ – the level at which odour recognition could be expected.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Wheeler et al (2000) suggested that downwind (8-150m) concentrations of respirable dust around 3 composting sites, monitored by Casella, were generally below a 1mg/m³ non-occupational exposure limit although no relationship could be made between concentration and distance. Turning produced the most dust. This study also suggests that odour levels downwind of the shredding, turning and screening operations varied between 35 and 6000U/m³ although measurements were complicated by upwind sources and no distances are given.

Comment on data quality:

Much of the quantitative data suffers from off-site distances of measurements not being quoted. The types of process and wastes being composted are also often not given.

Potential effects:

Both odour and dust may potentially be problems at composting operations. Odours are a particular cause of complaint and relate primarily to process failures. Dust is only likely to be a problem for sensitive receptors adjacent or very close to operations. Good housekeeping and process control should ensure that impacts are minimised. Nevertheless, in some situations, mitigation measures such as water and deodorant sprays may be necessary and enclosed systems may benefit from the use of biofilters or scrubbers.

Fauna

Number of relevant documents: 0

Summary of findings:

No Information.

Comment on data quality:

No information

Potential effects:

Potential effects on fauna due to composting operations are likely to derive from loss of habitat (which will depend on the location and size of the plant and the sensitivity of the habitat/species involved). Any releases to water may also potentially have a detrimental effect on aquatic species. Reductions in the use of peat based composts may also assist in protecting fauna associated with peat bogs.

Flora

Number of relevant documents: 1

Summary of findings:

Smith et al (2001) suggest that the primary ecological effect of compost production is to reduce the use of peat from peat bogs which are an increasingly rare habitat and harbour plants that favour waterlogged and acid conditions.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Comment on data quality:

Information is very limited and qualitative.

Potential effects:

Other than assisting in reducing demand for peat based products, flora may be affected by habitat loss during compost facility construction. The addition of compost will also enhance plant growth as long as contamination is avoided.

Soils

Number of relevant documents: 2

Summary of findings:

The Environment Agency (2002f) suggests that mixed waste compost will contain significant inert contamination such that compost derived from such wastes can only be used in the lowest quality applications (e.g. landfill cover). Heavy metal contamination is also an issue.

McLanaghan (2002) suggests that the application of finished compost is associated with improvements in soil structure, organic content, biological activity and fertility.

Comment on data quality:

Data are qualitative, limited and focus on the application of the finished compost rather than the impacts on soil from the process itself.

Potential effects:

The primary effect of compost on soils occurs during application, where the purpose of application is to improve soil quality, nutrient status and structure. Nevertheless, contamination may occur, particularly when using composts derived from MSW which may contain relatively high levels of heavy metals. Potential effects on soil due to the composting process could also arise from soil removal/compaction during the construction of the site and spillages of leachate resulting in some minor soil contamination.

Water quality

Number of relevant documents: 9

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that composting can create high organic strength leachates derived from high moisture content feedstocks. Leachate releases are most likely during the first 2 weeks of composting and could have deleterious effects on watercourses. Any such leachates should be recirculated into the compost and any excess collected and treated.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

The Environment Agency (2002f) states that leachate from composting can represent a potential hazard to surface or groundwater if accidentally released. However, mixed waste compost requires the moisture level to be maintained, such that any leachates are likely be recirculated. Composting in the open is likely to result in more leachate/contaminated run-off than enclosed systems and all operations need to be carried out on suitable hard surfaces to prevent groundwater contamination.

McLanaghan (2002) suggest that releases to water do not generally result from composting operations as leachate is generally recirculated or sent to a sewage works for treatment.

White et al (1995) suggests that aqueous effluents from composting processes vary greatly in volume and composition and that considerable evaporation will take place. Any collected water is often sprayed back onto the material being composted. Leachates tend to have elevated levels of biological oxygen demand, total organic carbon and ammonia.

The Environment Agency's 2001 report suggests that leachate from composting plants does not present any risk to the environment or public health.

Wheeler et al (2001) state that all process waters within the Lynbottom mixed waste composting site are re-used and, as such, there are no emissions to water.

Wheeler (2000) suggests that monitoring at 3 sites monitored by Casella indicated that leachate from composting activities was relatively low strength but may need treating before discharge.

Metcalfe et al (2000) suggest that compost leachate may contain chemical oxygen demand, chloride, magnesium, lead, nickel, zinc, cadmium, chromium, copper, mercury and cyanide.

Comment on data quality:

Information is primarily qualitative or relates to leachate concentrations.

Potential effects:

Leachates from composting facilities tend to be rich in organics, ammonia, nitrates and heavy metals giving them some polluting potential. However, leachates are generally produced in low volumes and tend to be recirculated within the compost process, or are treated. As such, the potential effects are limited to accidental releases. Any impact will depend on the volume and concentration of the release and the volume and quality/sensitivity/use of any receiving waters.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Water flow

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

The only effects on water flow likely to be associated with compost facilities will be due to increased run-off due to the presence of hard surfaces. The degree of any effect will depend on the area involved, local rainfall patterns and the capacity of the receiving watercourse. Adequate drainage will need to be provided. In general, as open schemes tend to take up more land per tonne of waste handled, open composting sites will have a greater potential impact on drainage. However, any effects will be no greater than for any other industrial facilities of similar size.

Air Quality

Number of relevant documents: 9

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that the primary atmospheric issue of concern at composting sites is the release of bioaerosols. Environment Agency research is quoted suggesting that bioaerosol levels tend to reach background levels within 250m of composting operations.

The Environment Agency (2002f) suggests that volatile organic compound emissions may be of concern and cites an Austrian study which reported emissions of benzene, toluene, xylene, trichloromethane and vinyl chloride (this study was carried out at an MBT plant). However, alkanes, aldehydes and alcohols were effectively controlled via biofilters. The report also suggests that bioaerosol emissions may also be of concern with windrow turning operations showing high emissions. Bioaerosol emissions are likely to be significantly lower for enclosed systems.

McLanaghan (2002) suggests that releases to air consist primarily of biogenic carbon dioxide and water along with bioaerosols, which will require monitoring within a 250m radius.

Tsiliyannis (1999) suggests that atmospheric emissions from composting plants may include carbon dioxide, ammonia and hydrogen sulphide.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

The Environment Agency (2001a) suggests that volatile organic compounds from composting operations are unlikely to present a risk to workers or the general public. Volatile readings were all very low or below the detection limit. Wide variations in readings were also noted. A wide range (57 species or genera) of airborne microbes were found in the air around the 3 composting sites monitored. Upwind and downwind concentrations ranged between the levels set out below (units are colony forming units per cubic metre – cfu/m^3):

	UPWIND:	DOWNWIND (20-40m):
Aspergillus fumigatus Total Fungi Total Bacteria Streptococci Enterobacteriaceae	$<10^{2} - 3 \times 10^{5}$ $<10^{3} - 5 \times 10^{5}$ $10^{3} - >10^{6}$ $0 - 10^{3}$ ND - 10 ⁵	$1 \times 10^{2} - 2 \times 10^{6}$ $1 \times 10^{3} - 2 \times 10^{6}$ $8 \times 10^{3} - 1 \times 10^{6}$ $0 - 3 \times 10^{4}$ $0 - 4 \times 10^{3}$
Total actinomycetes	10' - 10 [°]	$3 \times 10^{\circ} - 1 \times 10^{\circ}$

At Dogsthorpe and Morpeth downwind levels were between 1-2 orders of magnitude higher than upwind levels. At Netley, however, there was little difference. All downwind values are less that those adjacent to the plant. Typically, concentrations decreased by 80-90% between 20-40m from the source. Turning and shredding operations tend to liberate bioaerosols. The report also refers to a number of other (contradictory) studies which suggest that *A. fumigatus* (which is a class II pathogen) spores reach background levels between 75 - >1000m downwind of composting plants depending on weather conditions and operational parameters. The study concluded that further research was required to assess the risks to the general public from bioaerosol emissions. (NB. The types of operations and inputs at each site are not described).

Swan et al (2002) quote a number of studies measuring/modelling bio-aerosols both on and off-site at a range of composting plant and conclude that bioaerosol concentrations generally reach background levels within 100-500m from the site and may reach background levels within 250m of the compost. However, the study also concludes that different sampling methods and protocols can result in very different results. It also suggests that emissions can vary by up to 10-fold from hour to hour and that concentrations can vary widely depending on the type of composting activity, the weather, and compost moisture content.

Wheeler et al (2001) suggests that volatile organic compound concentrations in and around composting facilities are well below UK safety guidelines and are not believed to pose a threat to public health. During the study of 3 plants, 10 volatile organic compounds were recorded at levels above detection levels. It was found that the US ambient air quality limits were only likely to be exceeded for compounds such as benzene, toluene and xylene within 30m of operations. However, measured levels were very variable. The study also considered bioaerosols and suggested that concentrations within composting sites exceeded reference levels of 1000cfu/m³ (total bacteria), 1000cfu/m³ (total fungi) and 300cfu/m³ (gram -ve bacteria, which are associated with endotoxins in their cell walls which can lead to allergic reactions). Atmospheric levels varied between 10⁵ and 10⁶ cfu/m³ for bacteria and gramnegative bacteria and $10^3 - 10^4$ cfu/m³ for fungi. Modelling (whilst suffering from difficulties due to the influence of clumping of organisms and loss of organism viability over time) suggested that bioaerosol levels would reach the reference levels quoted within 250m distance.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Wheeler (2000) suggests that workers may potentially be affected by bioaerosols with on-site levels of *A. fumigatus* quoted between $10^3 - 10^7$ cfu/m3.

Metcalfe et al (2000) suggest that reported atmospheric emissions from composting facilities include: fungi, *A. fumigatus*, bacteria, streptococci, enterrobacteria, actinomycetes, xylenes, nonane, beta-pinene, ocimene, undecane, dodecane, methyl-(methylethyl)-cyclohexane, particulates, ammonia, water vapour and carbon dioxide

Comment on data quality:

Data often suffer from a lack of information regarding the distance from the source to the sampling point, weather conditions at the time and information on the process/waste type involved. Data appear to be limited to short term measurements.

Potential effects:

The above references suggest that volatile organic compound emissions are unlikely to present a problem around composting sites. There is, however, some concern regarding bioaerosols despite little firm guidance as to what acceptable levels are and at what levels health effects may arise. As such, sensitive siting is the preferred mitigation measure, with distances of 250m between the site and sensitive receptors often quoted.

Climate

Number of relevant documents: 3

Summary of findings:

The Environment Agency (2002f) suggests that carbon dioxide emissions from the composting process are all from 'short cycle' carbon and thus have little effect on climate change. However, emissions of the greenhouse gases methane and nitrous oxide could potentially occur at some sites. Monitoring suggests that, at well run sites, methane emissions are negligible as aerobic conditions are maintained.

Smith et al (2001) suggests that the greenhouse gas flux from open composting is around $-12kgCO_2eq/T$ and from closed composting is $-10kgCO_2eq/T$ taking all sources into account (including associated transport, the process itself, carbon sequestration in soils and displacement of emissions from peat use).

White et al (1995) suggests that composting a tonne of wet organic feedstock will liberate around 323 T of carbon dioxide.

Comment on data quality:

Data relies on a wide range of assumptions.

Potential effects:

Given that carbon dioxide emissions relate to 'short cycle' carbon, composting is unlikely to have a major effect on climate other than preventing the emission of methane (which is a much more potent greenhouse gas) from the landfilling of waste.

	Appendix 1		Appendix 2 Small Scale Incineration of Pre- Pyrolysis Sorted Wastes with Recovery Fneray				Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Acid gases

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Composting operations are not generally associated with emissions of acid gas and, as such, are unlikely to contribute to erosion of buildings. However, associated emissions of NO_x from vehicle movements may contribute to local NO_x levels.

Other environmental effects

Other environmental effects associated with compost facilities may include visual intrusion, litter, traffic and effects associated with the treatment and disposal of leachate and composted wastes.

	Appendix 1		Appendix 2 ed Incineration of Pre- Pyrolysis sorted Wastes with Energy Wastes with				Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.8 Mechanical Biological Treatment (MBT)

Noise

Number of relevant documents: 3

Summary of findings:

ODPM/Enviros (2003) (draft) suggests, in qualitative terms, that noise sources may include: on-site vehicle movements (including loading/unloading operations); associated traffic off-site; mechanical processes (including shredders, screens, trommels and ball mills); and noise from ventilation/fan systems. Noise limits at receptors are also suggested as 45-55dB(A) (daytime) and 35-45dB(A) (night-time) as for other industrial facilities.

Enviros (2003) report for Norfolk County Council relates, in part, to the Herhof MBT process and suggests that no noise issues generally arise as operations are enclosed within a building. However, noise may be associated with vehicles.

McLanaghan (2002) suggests that noise emissions are generally low-medium from MBT plant (similar to farm operations) but that shredding could result in higher levels.

Comment on data quality:

Quantitative data are not available. Qualitative comment is limited to primary sources of noise from MBT plants.

Potential Impacts:

Noise from MBT plants may arise primarily due to associated traffic noise with some contribution from mechanical operations. Noise impacts will depend upon the nature/intensity and level of enclosure of the operations and the distance/number of sensitive receptors. Off-site noise levels may be controlled by bunds/fences, sensitive siting, enclosure of noisy operations, regular plant maintenance and use of smart reversing alarms (i.e. those that limit their output to 5dB(A) above background) on vehicles etc. If such controls are utilised effectively noise nuisance is unlikely to arise.

Odour/dust

Number of relevant documents: 3

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that odours may arise at MBT plants simply due to the presence of putrescible wastes. Dust may also be derived from materials handling, especially if a drying process is used, and vehicle movements.

Enviros (2003) report to Norfolk County Council, relating to the Herhof plant, suggests that the use of a negative internal pressure controlled odours and that dust from the sieving of dried waste was contained and removed via a filtration process and, as such, implies that odour and dust impacts were minimal.

	Appendix 1		Appendix 2 Small Scale Incineration of Pre- tion of Pre- Pyrolysis Flaring				Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

McLanaghan (2002) suggests that odour levels from MBT plants tend to be low but that the exhaust process air requires odour control (e.g. via biofiltration).

Comment on data quality:

Quantitative data are not available. Qualitative comment is limited to primary sources and controls.

Potential Impacts

Odour/dust impacts will depend on proximity to, and number of, sensitive receptors, weather conditions and site orientation to receptors, the degree of enclosure of operations and the type and level of specific controls. Controls may include enclosure and use of negative air pressure, water sprays/bowsing, road sweeping, perfume sprays, dust filtration (cyclones, electrostatic precipitators etc.) odour filtration (biofilters, wet scrubbers, activated carbon, combustion etc.) and avoidance of waste materials on-site overnight. If such controls are applied effectively significant effects due to dust or odour are unlikely to arise.

Fauna

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

Data not available.

Potential Impacts:

Impacts on fauna are only likely to relate to the loss of habitat associated with the land-take of the plant, as at any other industrial facility of a similar scale.

Flora

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

Data not available.

Potential Impacts:

As for any other industrial facility of a similar scale, impacts on flora are only likely to relate to the loss of habitat associated with the land-take of the plant.

	Appendix 1		Appendix 2 Small Scale Incineration of Pre- sorted Wastes with Recovery Recovery				Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Soil

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

Data not available.

Potential Impacts:

Impacts on soil are only likely to relate to soil stripping associated with the landtake of the plant, as for any other industrial facility of a similar size. Hardstanding will help prevent soil contamination from waste residues.

Water quality

Number of relevant documents: 3

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that, due to the nature of the wastes being handled, there may be a risk to water resources from MBT plants. However, as most plants are under cover rain is unlikely to come into contact with waste materials and any risk is unlikely. Nevertheless, controls on washdown waters and liquids derived from the waste itself will be required.

McLanaghan (2002) suggests that MBT plants do not tend to result in liquid emissions to sewer. However, collection of liquid derived from waste could potentially be required with tankering off-site for treatment.

Hertfordshire (undated) makes similar suggestions to McLanaghan.

Comment on Data Quality

Quantitative data not available. Qualitative comment is limited to primary sources and controls.

Potential effects:

Contamination of water resources is unlikely in covered plants with adequate hardstanding. Drainage systems to sewer or to collection sumps for tankering to sewage plant may be required to prevent any dirty water from coming into contact with sensitive waters. Siting away from sensitive water resources may assist in reducing any risks.

	Appendix 1		Appendix 2 Small Scale Incineration of Pre- sorted Mathematical Scale inceration h Energy Wastes with				Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Water flows

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

Data are not available.

Potential Impacts:

Impacts on water flows may arise from the presence of hard surfaces (concrete surfaces and buildings) being located in previously soft surfaced areas (fields etc.) as with any industrial plant of a similar size. The larger the plant the more significant such effects could become, especially if located near water-bodies associated with flooding. Drainage systems may be required to ensure that such effects are minimised.

Air Quality

Number of relevant documents: 5

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that the primary atmospheric emissions associated with MBT are from associated vehicle movements along with bioaerosols and Volatile Organic Carbons (VOCs) from the biological treatment. It is suggested that atmospheric impacts may be similar to composting operations.

Enviros 2003 Report to Norfolk County Council suggests that the Herhof plant results in little impact on air quality due to stringent controls including enclosure of operations, collection of internal air and combustion of Volatile Organic Compounds/bioaerosols (most likely within an associated small scale incinerator/RDF plant). It does suggest, however, that the use of biofilters may allow the escape of some Volatile Organic Compounds.

Hertfordshire (undated) and McLanaghan 2002 indicate that atmospheric emissions are limited to harmless carbon dioxide and water thereby resulting in no air quality impact.

An Austrian study (Lahl et al, 1998) measured environmental concentrations of eight VOCs which could be emitted from Mechanical Biological Treatment plant. The study identified the following levels :

- ♦ Benzene 0.068 µg/m³
- Toluene
 0.82 μg/m³
- Ethylbenzene 2.8 μg/m³
- m/p-xylene $4.5 \mu g/m^3$

	Appendix 1	Appendix 2 Unsegregated Incineration with Energy Recovery Energy Energy					Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

- Styrene 0.46 µg/m³
- o-xylene $2.3 \,\mu g/m^3$
- Acetone 12.8 μg/m³
- 2-butanone 5 μg/m³

These levels are typical of urban and suburban levels of these substances, and do not indicate a significant increment due to the presence of the MBT facility.

Comment on data quality:

Quantitative measurements of a limited dataset of VOCs are available. Qualitative comment is otherwise limited to primary sources and controls.

Potential Impacts:

Air quality impacts due to MBT are unlikely to be significant. Some emissions of VOCs and bioaerosols may occur but are unlikely to affect air quality. Similarly, traffic emissions are unlikely to be significant unless the plant is located in a sensitive area such as a Local Authority Air Quality Management Area (AQMA) where air quality is already poor. Potential controls may include sensitive routing of vehicles and provision of Volatile Organic Compound/bioaerosol controls (e.g. combustion, biofiltration, scrubbing etc. along with internal negative pressures within buildings). Combustion emissions may be derived from any associated small scale incineration/RDF plant (see Small Scale Incineration).

Climate

Number of relevant documents: 1

Summary of findings:

Smith et al 2001 provides a very detailed analysis of greenhouse gas emissions from a range of waste management options and combinations of option scenarios (including MBT) taking account of greenhouse emissions throughout the waste stream (including transport, treatment, disposal of residues, emissions avoided from fossil fuel power production etc). The MBT process in isolation (i.e. without consideration of transport, residue disposal or emissions avoided) results in emissions of $22kgCO_2eq/tonne$ of MSW treated.

Taking all emissions and emissions savings into account MBT with landfill of residuals results in a negative greenhouse gas flux of around -340kgCO₂eq/tonne whereas, with incineration of residuals, there is a negative flux of around -230kgCO₂eq/tonne. MBT provides the lowest greenhouse flux of all waste treatment options prior to landfilling. This is primarily due to the reduction of methane production potential from the composting of putrescible wastes prior to landfilling. However, the efficiency of landfill gas controls and local circumstances will affect the overall greenhouse gas efficiency of the overall waste management scheme. Further details are provided in section 3.1.4 and Appendix 4 of the reference.

	Appendix 1		Appendix 2 Small Scale Incineration of Pre- sorted Wates with Energy and/or				Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Comment on data quality:

Data provided in Smith et al are very comprehensive, but necessarily rely on a number of assumptions and options including, for example, location of plant on a landfill site, assumed transport distances and average European use of power plant.

Potential Impacts:

The use of MBT as pre-treatment of waste prior to final disposal is likely to result in net overall reductions in greenhouse emissions from waste management activities. Actual greenhouse performance will depend on local circumstances and the precise nature of the processes involved.

Acid gases

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

Data are not available.

Potential Impacts:

Given that atmospheric emissions from the MBT process are limited to carbon dioxide and water (see Air Quality section above) the process *per se* is unlikely to have any effect on acid gas levels and therefore building erosion. Associated traffic emissions of NOx may, however, contribute to the concentrations of acid gases in local air. Combustion emissions of acid gases may also be derived from any associated small scale incineration/RDF plant (see Small Scale Incineration section).

Other Potential Environmental Effects

Other effects may include visual intrusion, litter, traffic and residual disposal issues.

	Appendix 1		Appendix 2 Small Scale Incineration of Pre- Pyrolysis Landfill With Landfill Gas Flaring			Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.9 Materials Recycling Facilities

Noise

Number of relevant documents: 4

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that noise may be emitted from MRFs handling source separated recyclables ('clean MRFs') from vehicle manoeuvring, traffic, mechanical plant (including shredders, screens, conveyors, trommels and crushers) and ventilation systems. Noise limits at off-site receptors are typically 45-55dB(A) (daytime) and 35-45dB(A) (night-time), similar to other industrial facilities. For ODPM/Enviros (2003) comments on 'dirty MRFs' see section on MBT.

The Environment Agency's (2002) report on waste pre-treatment suggests that complaints are unlikely if the MRF is situated at a landfill or in an industrial area. However, traffic noise may be an issue.

McLanaghan (2002) suggests that noise levels associated with MRFs are 'lowmedium' and similar to farm operations. However, the more automated plants may require greater noise control.

Weston (1995) provides instantaneous noise monitoring data for 6 clean MRFs in the USA both on the site peripheries and at off-site locations. The ranges of results for each site are given below:

Islip MRF	Fence line = $46-76dB(A)$
	Off-site = $51-74dB(A)$

The main source of noise was glass and other materials being dumped on the floor and associated truck movements.

<u>Montgomery MRF</u> Off-site = 74dB(A)

The primary noise sources were the process line and tub grinder. However, the site is remote from noise sensitive land uses and therefore has no significant noise impact. (NB. The MRF is adjacent to a waste transfer station).

<u>Albuquerque MRF</u> Off-site = 55-70dB(A)

The main noise source was glass crushing along with trucks. However, the site is remote and no significant impacts occur. (NB. The MRF is adjacent to a landfill).

<u>Hartford MRF</u> Off-site = 57-73dB(A)

(NB. The MRF is adjacent to a RDF plant).

	Appendix 1 Appendix 2 Unsegregated Incineration Statements Unsegregated Incineration Statements Unsegregated Incineration Vastes						Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Rice County MRF

Fence line = 50-58dB(A)Off-site = 48-66dB(A)

The site is in a remote farming area and is unlikely to cause any noise nuisance. (NB. The MRF is close to a hazardous waste storage area).

<u>Orange County MRF</u> Fence line = 59-74dB(A)

The site is in a remote location and does not cause a noise impact. (NB. The MRF is close to a landfill and hazardous waste station).

Comment on data quality:

Most references refer to noise sources and controls. However, Weston (1995) reports measured noise levels at perimeter and off-site locations. The value of this information, however, is reduced by a lack of provision of distances between source and receptors (locations of monitoring points are marked on unscaled maps), a lack of background data without the plant being operational (for comparison) and the use of instantaneous noise measurements rather than time-weighted averages (as normally used in the UK). Such omissions make the data difficult to interpret in terms of UK standards and the likelihood of complaints. In addition, many of the MRFs monitored are located very close to other waste management operations which could have affected the results. However, from the text it is clear that none of the MRFs monitored resulted in community annoyance due to their isolation from sensitive receptors. The paper states that community noise levels met applicable federal and state criteria.

Potential effects:

Noise may be a potential issue at MRFs as certain operations, such as glass handling, can be particularly noisy and, given their intermittent nature, can be especially annoying. As such, sensitive siting is important and other controls, such as the use of soundproofing, bunds, fences and 'smart' reversing alarms (i.e. those which automatically limit their output to 5dB(A) above background) may need to be considered.

Odour/dust

Number of relevant documents: 3

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that odour at clean MRFs tends to be limited to that derived from residual liquid within bottles and materials contaminated with residual biodegradable matter. Waste handling and vehicle movements may give rise to dust. Such emissions are relatively easy to control. For ODPM/Enviros (2003) comments on 'dirty MRFs' see section on MBT.

The Environment Agency (2002f) suggests that odours should not be an issue at clean MRFs. Dirty MRFs may be more problematical as they accept unsorted waste. However, this may be overcome by sensitive siting and the application of controls. Dusts may also be controlled effectively at either type of MRF via effective ventilation. However, bioaerosols may potentially be of some concern, especially at dirty MRFs.

	Appendix 1	Appendix 2 Unsegregated Unsegregated Uncineration Unsegregated Uncineration Unsegregated Uncineration Unsegregated Uncineration Unsegregated Unsegregated Unsegregated Unsegregated Unsegregated Unsegregated Unsegregate Un					Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

McLanaghan (2002) suggests that, for clean MRFs, odour levels should be 'very low'. At dirty MRFs odour levels should be 'low-medium' at the site perimeter and that the use of biofilters may assist in minimising odour releases.

Comment on data quality:

Information is qualitative and focuses on sources and controls.

Potential effects:

Odour effects are only likely at dirty MRFs (also see section on MBT). At such sites sensitive siting should be considered along with effective ventilation and deodorant sprays. The use of air extraction and treatment using techniques such as biofiltration may be useful at dirty MRFs.

Fauna

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Effects on fauna are likely to be limited to habitat loss through facility construction. As such, any effects are likely to be equivalent to those for any similar sized industrial facilities.

Flora

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

As for fauna, effects on flora are likely to be limited to habitat loss. As such, any effects are likely to be equivalent to those for any similar sized industrial facilities.

	Appendix 1			Appendix 2			Appendix 3			Appendix 4	
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Soils

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Any effects on soils are unlikely except due to soil removal during construction. As such, any effects are likely to be equivalent to those for any similar sized industrial facilities. There may also be a possibility of minor localised soil contamination at dirty MRFs if waste or leachates are released to the wider environment.

Water quality

Number of relevant documents: 4

Summary of findings:

ODPM/Enviros (2003) (draft) states that residual liquids in bottles and cans could potentially pose a risk to water resources. However, as most facilities are under cover and on hardstanding any such risk is low. Nevertheless, wash-down waters and any liquid within the waste will need to be handled appropriately. For ODPM/Enviros (2003) comments on 'dirty MRFs' see section on MBT.

The Environment Agency (2002f) suggests that clean MRFs should not be associated with water pollution. Dirty MRFs, however, may potentially produce contaminated leachate from organic wastes. Any such leachate will require collection and treatment.

McLanaghan (2002) states that MRFs should not discharge liquids into the sewerage system. However, dirty MRFs should be designed to collect any liquid which should then be tankered off-site for treatment.

Weston et al (1995) suggests that washdown waters (where they occur) may contain elevated levels of chemical oxygen demand, ammonia, total organic nitrogen, total organic carbon, oil and grease, phosphate, conductivity, total suspended solids, total dissolved solids, biological oxygen demand, silver, arsenic, barium, cadmium, chromium, mercury, lead, selenium, total and faecal coliforms. However, many facilities do not create any contaminated water.

Comment on data quality:

Information is either qualitative and focused on controls, or is quantitative but limited to concentration data.

	Appendix 1			Appendix 2		Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

Given the low or non-existent volumes of contaminated water produced by most MRFs water pollution effects are unlikely. Releases from dirty MRFs could potentially have an effect on very small receiving watercourses. However, appropriate collection and treatment of contaminated waters will prevent any such impacts. Bunding of oil/fuel stores is also required.

Water flow

Number of relevant documents: 0

Summary of findings:

No information available.

Comment on data quality:

No information available.

Potential effects:

The only potential effect on water flow due to the presence of a MRF will be due to increased run-off from hard surfaces such as buildings and hardstanding. The nature of the effect will depend on the land area involved, the volume of the receiving watercourse and local weather patterns. Adequate drainage systems are required. However, any effect is likely to be equivalent to any other similar sized industrial facility.

Air Quality

Number of relevant documents: 4

Summary of findings:

Lavoie and Guertin (2001) is a Canadian paper relating to bioaerosols, dusts, carbon monoxide, NOx, noise, vibration, lighting, magnetic fields and ergonomics in relation to worker health at recycling plants. Primarily internal levels given. However, bioaerosol levels are also given 300m upwind and 100m downwind for summer and winter. Summer downwind ranges were as follows:

Total Bacteria = 520-5650 colony forming units per cubic metre (cfu/m^3);

Gram negative bacteria (which have toxins in their cell walls which can lead to allergic reactions) = ND-250cfu/ m^3 ,

Moulds = 730-3095 cfu/m³.

In comparison to Scandinavian guidelines of 10,000cfu/m³ total bacteria and 1000cfu/m³ Gram negative bacteria such levels appear not to be significant. (NB. Benchmarks from literature associated with composting are often quoted as 1000cfu/m³ for total bacteria or fungi and 300cfu/m³ for Gram negative bacteria).

ODPM/Enviros (2003) (draft) suggests that atmospheric emissions associated with clean MRFs are dominated by vehicle emissions. Due to the absence of

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

degradable wastes, air quality is unlikely to be an issue. For ODPM/Enviros (2003) comments on 'dirty MRFs' see section on MBT.

McLanaghan (2002) states that MRFs have no regulated emissions to air.

Weston et al (1995) took air measurements upwind and downwind of 6 MRFs in the USA. Determinands covered included total suspended particulates, PM_{10} , lead, carbon monoxide, mercury, volatile organic compounds (various), and bacteria/fungi. In general terms no determinands exceeded local or national standards. There was no significant difference between upwind and downwind lead levels at any site and carbon monoxide / mercury levels were either not detectable, showed no difference between upwind and downwind sites and, where detected, were not above expected background levels. No particularly virulent pathogens were noted at any of the sites. The other results for each of the sites are summarised below:

Islip MRF (clean)

Total suspended particulate levels showed a potential downwind increase on 2 of the 3 days monitoring increasing from 40 or 38 to $61\mu g/m^3$ and from 66 to $95\mu g/m^3$ respectively.

 PM_{10} showed similar results increasing from 25 to 28 or $37\mu g/m^3$ on one day and from 33 to 46 or $75.5\mu g/m^3$ on another day.

Volatile organic compounds showed no significant difference between up and downwind locations and levels were typical of background.

Bacteria and fungi levels were an order of magnitude lower at external locations than inside the MRF. Downwind and upwind levels showed no significant difference suggesting that the MRF was not emitting microbes significantly.

Montgomery County MRF (clean but adjacent to a waste transfer station)

Total suspended particulate levels showed increases in downwind levels on 2 of the 3 sampling days increasing from 2 to 85 or $146\mu g/m^3$ on one day and from 64 to 134 or $323\mu g/m^3$ on another.

 PM_{10} levels were more variable with upwind concentrations between 38 and $60\mu g/m^3$ and downwind concentrations ranging between 39 and $335\mu g/m^3$. On 2 of the 3 days downwind concentrations were both above and below upwind concentrations depending on the sampling location.

As at Islip volatile organic compound concentrations were typical of background and showed no significant difference between upwind and downwind locations.

Indoor fungi were 2-3 times higher than outside except on one day when levels were roughly equivalent. No obvious differences were apparent between up and downwind locations.

Albuquerque MRF (clean but adjacent to a landfill)

Total suspended particulate levels on 2 of the 3 days increased in downwind locations from 17.5 to $63.4\mu g/m^3$ and from 65.4 to $92\mu g/m^3$ respectively. PM₁₀ levels showed no particularly significant change.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

No volatile organic compounds were detected except acetone which showed its highest concentration upwind.

Bacteria and fungi were 1-2 orders of magnitude lower outdoors than indoors suggesting that microbes are not released in measurable quantities.

Hartford MRF (clean but close to RDF plant)

Total suspended particulate and PM_{10} results indicated that a paper recycling facility caused negligible contributions to perimeter levels. However, a container recycling operation resulted in moderate contributions to perimeter particulate and PM_{10} concentrations. Total suspended particulate levels increased from 46 upwind to 139µg/m³ downwind whereas PM_{10} increased from 29 to 52.4µg/m³.

Volatile organic compounds showed a very slight increase downwind. Acetone increased from 23 downwind to 31 or $33\mu g/m^3$ upwind. Benzene climbed from 1.9 to $2.2\mu g/m^3$, toluene increased from 9.4 to 11 or $41.4\mu g/m^3$, 1.1.1 trichloroethane increased from ND to $1.1\mu g/m^3$ and xylenes increased from ND to $6.9\mu g/m^3$.

Fungi and bacteria were an order of magnitude more concentrated within the building than outside and downwind levels were no higher than upwind indicating no significant release.

Rice County MRF (clean but adjacent to a landfill and a hazardous waste storage area)

Total suspended particulate levels were higher upwind on 2 days out of 3. The only downwind increase was from 9.7 to 16.5 or $27.2\mu g/m^3$.

Downwind PM_{10} levels also showed no major increase except on 1 day when levels increased from 15.8 to 28.8 and $89.8\mu g/m^3$.

There was no significant difference between up and downwind volatile organic compound levels except in the case of toluene which rose from 1.5 to 10.2 or $21.1\mu g/m^3$.

Bacterial and fungal levels were 1 to 2 orders of magnitude lower outside than inside. Bacteria showed higher levels upwind indicating a possible alternative source. Only one sample showed an increase in fungi downwind (being approximately double that upwind).

Orange County MRF (clean but adjacent to a landfill and hazardous waste station)

Total suspended particulate levels downwind appeared elevated against upwind locations on 2 out of 3 days sampling increasing from 38 to 62 or $71\mu g/m^3$ on one day and from 43.5 to 54.5 or $111\mu g/m^3$ on another.

 PM_{10} levels showed a similar pattern increasing from 18-29µg/m³ on one day and from 21 to 23 or 47 µg/m³ on the other.

Volatile organic compounds showed no difference between up and downwind samples and levels were representative of normal background concentrations.

Bacteria and fungi were 1-2 orders of magnitude less concentrated outside than inside. However, on 1 out of the 3 days sampling, downwind concentrations were significantly higher than upwind, suggesting the MRF was the source. Fungi

	Appendix 1			Appendix 2		Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

increased from 220-440 to 348-3437cfu/m 3 and bacteria increased from 209-1136 to 336-2730cfu/m $^{3}.$

Comment on data quality:

Much of the available information is qualitative and focuses on controls. Weston et al however, reports measured off-site air quality. However, the value of this information is reduced by distances of sampling locations not being given, a lack of clarity as to which samples are up or downwind and the 3 day sampling period cannot be representative of long term effects. In addition, many of the MRFs are close to other waste management operations which could affect the results.

Potential effects:

Whilst from the qualitative literature, it appears that MRFs are unlikely to create significant impacts on air quality, the Weston study suggests that elevated levels of particulates may be found on site peripheries and in some cases elevated levels of microbes may occasionally occur. However, the short term nature of this study and the great variation in results (especially where several particulate results upwind are higher than downwind) suggests that background variability may be more important. Nevertheless, the data from Weston et al suggest that no standards were breached and, as all sampling locations were close to the site peripheries, this would suggest that no significant impacts on air quality are likely. However, dust controls could potentially be useful to ensure that risks are minimised.

Climate

Number of relevant documents: 2

Summary of findings:

The Environment Agency (2002f) suggests that the operation of a clean MRF is generally a low energy process whereas dirty MRFs utilise more energy. Material export also adds to transport related emissions that may otherwise not occur due to waste collection and transport.

Smith et al (2001) provides greenhouse gas fluxes for MRF operations (including transport to and from the MRF per tonne of material recycled as follows:

Paper/glass/ferrous metals/textiles/aluminium = 10.31 kgCO₂eq/T Plastic = 15.29 kgCO₂eq/T WEEE = 12.08 kgCO₂eq/T

Comment on data quality:

Smith et al relies on a range of assumptions and The Environment Agency's document does not explain how conclusions were derived.

Potential effects:

Given that no power is exported from MRFs and transport to recycling facilities is often long distance, MRFs are likely to result in a slight negative effect on greenhouse emissions.

	Appendix 1			Appendix 2		Appendix 3			Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Acid gases

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

MRF operations are not associated with acid gas emissions and will thus not result in building erosion. Emissions of NOx from associated vehicles, however, could potentially contribute to local ambient NOx levels.

Other environmental effects

Other environmental effect may be associated with visual intrusion, litter, further recyclate processing and fuel use.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.10 Waste Transportation

No research specific to the transportation of waste could be found. The environmental effects of transportation will be similar to those for transportation of other industrial products. Transportation of MSW accounts for approximately 0.5% of heavy goods vehicle mileage in the UK. Some MSW is also transported by train or boat, but this represents a small proportion of MSW movements in the UK.

This section therefore focuses on information relating to waste transfer stations, which form a part of the process of transportation of waste.

Noise

Number of relevant documents: 1

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that noise problems at waste transfer stations have been attributed to vehicle manoeuvring (especially vehicle reversing alarms) and that such operations can be particularly noisy in comparison to other waste management operations. Noise may also derive from vehicles on the local road network. Typical limit values are 45-55dB(A) (daytime) and 35-45dB(A) (night-time) in line with other industrial processes.

Comment on data quality:

Information very limited and qualitative.

Potential effects:

Traffic and vehicle manoeuvring may be problematic if transfer stations and associated routes are located in, or near, noise sensitive land uses (e.g. residential areas). Waste vehicles should be routed away from sensitive areas, operational hours may be limited and smart reversing alarms (i.e. those which limit their output to 5dB(A) above background) could be used to minimise risks of noise nuisance.

Odour/dust

Number of relevant documents: 1

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that the presence of putrescible wastes in a transfer station may lead to detectable odour nearby. However, fast turn around times usually prevent any serious problem. Dusts may also be produced from vehicle movements and waste handling but transfer stations are not normally associated with dust nuisance.

Comment on data quality:

Information is limited and qualitative.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

Minor odour and dust impacts may be possible at waste transfer stations given the nature of the material being handled and the number of vehicle movements. Enclosure within a building is the primary means by which such impacts may be prevented along with water and perfume sprays.

Fauna

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information

Potential effects:

Potential effects of waste transfer on fauna are likely to be limited to habitat loss from the building of transfer stations, as with any industrial activity of a similar scale.

Flora

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information

Potential effects:

Like any other industrial activity potential effects of waste transfer on flora are likely to be limited to habitat loss from the building of the transfer station.

Soil

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

Potential effects of waste transfer on soils are likely to be limited to soil stripping during transfer station construction and potential minor contamination from dirty run-off. Contamination may be prevented by the use of adequate drainage systems and hardstanding.

Water quality

Number of relevant documents: 1

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that, given the nature of the material being handled, there may be a risk of contamination of water resources. However, as transfer stations are primarily under cover, rain is unlikely to come into contact with the waste, thereby reducing such risks. Nevertheless, washdown waters and any liquid released directly from the waste will need appropriate handling via a drainage system to ensure that pollution is avoided.

Comment on data quality:

Information is limited and qualitative.

Potential effects:

Contamination of water is possible from washdown and liquids released from the waste itself. Drainage systems are required to separate dirty and clean water within transfer stations. Other potential effects may result from fuel/lubrication spillages from associated vehicles. Fuel stores should be bunded. The effect of any release will depend on both the nature/volume of the release and the volume/character of the receiving water body.

Water flow

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Water flows may be altered due to the presence of hard surfaces increasing run-off volumes. As such, drainage systems may be required. The effect of increased flows will depend on the volume of the receiving water-body but given the relatively small size of waste transfer stations flooding is unlikely.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Air Quality

Number of relevant documents: 2

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that emissions of combustion products (COx, SOx, NOx, volatile organic compounds and PM_{10}) will arise from HGV movements and that such emissions may have an effect on local air quality along the routes to and from the transfer station. However, on a regional scale the use of transfer stations reduces the overall HGV mileage travelled and, therefore, fuel use and emissions.

White et al (1995) suggests that atmospheric emissions from waste transfer will depend on the distances travelled and the volumes of waste handled.

Comment on data quality:

Information is limited and qualitative.

Potential effects:

Air quality impacts are limited primarily to those related to vehicle emissions. These are only likely to be a problem very close to the roads used for haulage and if existing air quality is poor (i.e. in an Air Quality Management Area (AQMA)). Bioaerosols may also be an issue, although to a much more limited extent than for composting operations.

Climate

Number of relevant documents: 1

Summary of findings:

Smith et al (2001) provides estimated figures for greenhouse emissions from waste transfer station to disposal/management options using assumed mileages and payloads as well as direct transfer figures. Figures in carbon dioxide equivalents emitted per tonne of waste transferred are provided as follows:

Direct to landfill or incinerator = 4.3kgCO₂eq/T (assumed 40km + 6.67 T payload) Collection to transfer station = 4.3kgCO₂eq/T (assumed 40km + 6.67 T payload) From transfer station to landfill or incinerator = 3.6CO₂eq/T (assumed 40km + 5 T payload).

Comment on data quality:

The assumptions used to provide the data given in Smith et al (2001) appear dubious as the purpose of using waste transfer stations is to reduce vehicle mileage by bulking waste up. As such, the assumption that payloads drop from 6.67 T to 5 T when transferring waste out of a transfer station appears unlikely. Using the same distance for all options to derive emission factors is also dubious as high numbers of low capacity collection vehicles will travel short distances to the transfer station whereas small numbers of high capacity transfer vehicles will travel longer distances to the final disposal/management facility.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Potential effects:

The purpose of waste transfer stations is to reduce the mileage, fuel use and therefore emissions from HGVs. Therefore, it is likely that, whilst transfer will contribute carbon dioxide from vehicle emissions to the greenhouse effect, the use of transfer stations will reduce the net effect to less than that which would occur if waste was transferred direct.

Acid gases

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Vehicles will emit NOx which could potentially contribute to local concentrations of acid gases. However, such contributions are unlikely to result in a measurable effect on building erosion.

Other Effects Noted

ODPM/Enviros (2003) (draft) suggests that waste transfer may also be associated with environmental impacts related to traffic, flies and vermin, litter and visual intrusion.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
- Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.11 Anaerobic Digestion with Energy Recovery

Noise

Number of relevant documents: 4

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that noise emissions are similar to other waste plants. Noise is primarily associated with vehicle movements and loading/unloading operations. Engines and pumps may also be particular noise sources. Typical noise limits at receptors are given as 45-55dB(A) (daytime) and 35-45dB(A) (night-time) similar to other industrial activities.

The Environment Agency (2002f) highlights the benefits of enclosure of activities within buildings to limit noise from shredding and processing operations. Problems may be caused by noise from fans and pumps at night when background noise levels drop. However, the primary source of noise at anaerobic digestion sites is usually the biogas electricity generator.

Enviros' Report for Norfolk County Council (2003) suggests that the German Biotechnische Abfallverwertung (BTA) anaerobic digestion process results in minimal noise problems and that noise is primarily a result of vehicle movements, as at other waste management operations.

McLanaghan 2002 suggests that noise emissions from anaerobic digestion plants are generally 'low-medium' and similar to a farm operation.

Comment on data quality:

Information is primarily qualitative and relates mostly to sources and controls.

Potential effects:

Like all waste management options noise is emitted from anaerobic digestion plants primarily in relation to vehicle movements. 24hr operations including noise from fans, pumps and engines may be problematic if the site is located close to sensitive receptors, such as residences. As such, noise controls may be required including, acoustic enclosures, generator exhaust silencing, bunds, fences and 'smart' reversing alarms (ie. those that limit their output to 5dB(A) above background levels) for vehicles etc.

Odour/dust

Number of relevant documents: 4

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that odour is perceived as the primary planning issue in relation to Anaerobic Digestion. However, the enclosure of operations within a building reduces the odour risk. Dust may also be produced from the loading/unloading of vehicles.

	Appendix 1			Appendix 2			Appendix 3		Appendix 4		
- Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

The Environment Agency (2002f) suggests that odours primarily derive from feedstock processing and digestate treatment. However, if these operations are enclosed within a building with appropriate controls (e.g. negative pressure and air fed to a biofilter, chemical scrubber or biogas combustion) there should be no problems. Suitably controlled plants in Europe have been located within industrial estates and have attracted no odour complaints from neighbours. Indeed some are as little as <10m from the nearest neighbour.

Enviros report to Norfolk County Council (2003) suggests that odour and dust may arise in the waste delivery/sorting areas but that the use of negative internal pressures, biofilters and good housekeeping may control odour release.

McLanaghan (2002) suggests that odour from anaerobic digestion plants is generally low due to the enclosed nature of the process. Ammonia odours may arise from the aeration stage of digestate treatment.

Comment on data quality:

Information is non-quantitative and relates primarily to odour/dust sources and controls.

Potential effects:

Odours may be released from anaerobic digestion plants primarily from feedstock and digestate handling. However, serious odour problems are usually avoided by the enclosure of operations within a building and the use of controls such as biofilters or scrubbers or the combustion of internal air with the biogas. Dust is not normally associated with anaerobic digestion plants.

Fauna

Number of relevant documents: 1

Summary of findings:

ODPM/Enviros (2003) (draft) suggests that, as waste-water may be produced from anaerobic digestion containing high concentrations of metals, dissolved nitrogen and organic matter, there is some potential for local ecosystem damage if an accidental spillage were to occur.

Comment on data quality:

Very little information available. A qualitative evaluation suggests that the only potential impact may be from accidental releases.

Potential effects:

Potential effects of the anaerobic digestion process on fauna are from rare accidental spillages, and habitat loss due to the land-take associated with the plant (as with any other industrial facility of a similar scale). Also the application of the resulting compost to land may potentially result in contamination issues and more subtle effects on local ecology. Equally, the improvements to primary production derived from compost application may result in greater food availability to animals.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
- Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Flora

Number of relevant documents: As for fauna.

Summary of findings:

As for fauna.

Comment on data quality:

As for fauna.

Potential effects:

As for fauna. Also plant growth is likely to be enhanced by the application of compost.

Soil

Number of relevant documents: 1

Summary of findings:

The Environment Agency's report (2002) suggests that there may be some risk of soil contamination with heavy metals or other substances when compost from anaerobic digestion is applied. This is especially the case when mixed wastes are utilised as a feedstock. Nevertheless, the application of such compost to soil is likely to result in improved water retention, improved soil structure, increased microbial activity and enhancement of the effect of inorganic fertilisers due to the presence of organic matter within the compost.

Comment on data quality:

Information limited and primarily related to the application of compost, resulting from anaerobic digestion, to soils as a fertiliser/soil improver.

Potential effects:

In general, the application of composts from anaerobic digestion to soils results in a positive impact to soil structure and performance as a growing medium. However, contaminant loadings and application rates need to be controlled/monitored to ensure that soils do not become contaminated with excessive levels of heavy metals etc. No other significant impacts on soils are likely to occur due to the presence of an anaerobic digestion plant, barring the removal of soils during construction and, potentially, contamination from spillages of leachate.

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
- Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

Water quality

Number of relevant documents: 7

Summary of findings:

ODPM/Enviros (2003) (draft) indicates that dewatering of digestate produces a liquor with high heavy metal, dissolved nitrogen and organic matter concentrations which may result in pollution if released untreated. As such, treatment is required either on-site or at a sewage treatment works. All liquor stores should be contained within bunded areas.

The 2002 Environment Agency report suggests that anaerobic digestion of a tonne of waste may produce 100-330kg of liquor which will require treatment either on or off site.

McLanaghan (2002) suggests that the liquor may be utilized as a fertilizer (but this is only currently practiced in Denmark) and the only releases to water are to sewer.

Hertfordshire (undated) suggests that anaerobic digestion results in minimal wastewater emissions as the process is a very small net water user. However, small volumes are treated via sewage works.

White et al (1995) suggests that liquor is produced when digested material is pressed or filtered. Some is recycled and the rest is treated prior to discharge. In general, between 290-500l/T of waste is produced. Concentrations of contaminants range as follows: biological oxygen demand = 60-740mg/l, chemical oxygen demand = <250-1400mg/l, ammonia = <100-250mg/l, total nitrogen = 6-<100mg/l, pH = 8.

Metcalfe et al (2000) suggest that anaerobic press water has elevated levels of dissolved solids, nitrogen, ammonia, nitrite, nitrate, chemical oxygen demand and biological oxygen demand.

Older information from Dasgupta et al (1981) primarily deals with contents of inputs and outputs of the anaerobic digestion process including ammonia, nitrite, nitrate, total organic nitrogen, phosphorus, chloride, sulphide, biological oxygen demand, chemical oxygen demand, total organic carbon, solids, faecal coliforms and heavy metals. This information relates mostly to emissions. One section does, however, relate to the potential effects of effluent on water quality particularly in relation to drinking water. This highlights a range of potential health and environmental hazards which could arise if filtrate were to be discharged untreated.

Comment on data quality:

Information is primarily qualitative except in terms of emission volumes and contaminant concentrations. Most references are very general. Dasgupta et al (1981) is more specific in relation to potential effects that could arise in the event of an accidental release of untreated liquid effluent.

Potential effects:

In general, liquor releases from anaerobic digestion plants are treated on-site, or via the sewerage system, prior to release to the environment, thereby resulting in

Appendix 1			Appendix 2			Appendix 3			Appendix 4		
- Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

minimal impact to water quality. Nevertheless, accidental releases could result in significant pollution particularly given the high ammonia and organic strengths of the liquor produced. The degree of pollution will depend entirely upon the volume and strength of the liquor released and the volume/quality of the water-body receiving the discharge (i.e. a high volume discharge of high strength liquor to a small pond could be devastating whereas the same release to a large river with a high organic and nutrient content may have virtually no effect).

Water flow

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

Potential effects are likely to be limited to the replacement of soils with impermeable concrete/building structures when the plant is built, resulting in increased run-off. Such impacts will depend on the plant area, local rainfall patterns and the volume of receiving watercourses. Appropriate drainage arrangements will be required. However, any effect will be equivalent to any similar sized industrial operation.

Air Quality

Number of relevant documents: 6

Summary of findings:

White et al (1995), Hertfordshire (undated) and McLanaghan (2002) suggest that the emissions from anaerobic digestion comprise carbon dioxide and water with traces of acid gases such as sulphur dioxide.

Enviros 2003 report for Norfolk County Council indicates that atmospheric emissions are minimal.

The Environment Agency (2002f) suggests that the primary emissions to air from anaerobic digestion derive from biogas combustion resulting in emissions of NOx and SOx in similar proportions to the burning of natural gas (although SOx levels may be higher due to the presence of hydrogen sulphide). However, such emissions may be off-set against emissions from fossil fuel use avoided.
Appendix 1			Appendix 2		Appendix 3		Appendix 4				
- Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

ODPM/Enviros (2003) (draft) indicates that emissions data are very limited but that anaerobic digestion has relatively low atmospheric emissions in comparison to other waste management options. However, despite enclosure of the process some fugitive emissions may be possible, along with the emission of bioaerosols.

Comment on data quality:

Information is primarily qualitative and relating to emissions rather than effects. Many sources suggest that carbon dioxide and water are the only major emissions. However, this is erroneous as combustion of any fuel (including biogas) will result in the emission of NOx via the oxidation of nitrogen in the combustion air and other combustion products (e.g. volatile organic compounds and SOx).

Potential effects:

As emissions are low and primarily from biogas combustion, which can be off-set against emissions avoided from fossil fuel combustion, anaerobic digestion is unlikely to have significant effects on air quality. However, biogas combustion and emissions from associated traffic could potentially contribute to local levels of NOx etc.

Climate

Number of relevant documents: 4

Summary of findings:

McLanaghan (2002) suggests that combustion of biogas displaces greenhouse emissions from fossil fuel use and that the carbon locked up in the digestate compost is only released to atmosphere over decades, thereby prolonging short term carbon cycling.

Smith et al (2001) provides a number of greenhouse gas emission factors for waste transport, the process itself and emissions avoided from fossil fuels. However, no actual overall figures are given for MSW as greenhouse emissions depend on the amount of putrescibles/paper within the waste. Nevertheless, it suggests that fugitive emissions from the treatment process could be around 0-10kgCO₂ eq/T and that any fugitive emission of methane could significantly alter the process greenhouse gas flux as methane is a relatively powerful greenhouse gas. The report also indicates that the use of the resulting compost sequesters carbon in the soil. Given emissions avoided from fossil fuel use, a generally negative greenhouse gas flux results from anaerobic digestion. Indeed, the paper suggests that taking all emissions into account from the transport and digestion of 'putrescible' waste along with emissions foregone from power production and soil carbon sequestration, fluxes of -33 and -58kgCO₂eq/T for anaerobic digestion with power production and CHP may be achieved respectively.

The Environment Agency (2002f) suggests that all carbon dioxide emitted from anaerobic digestion is 'short cycle' and thus, does not have a significant effect on climate change. However, any fugitive emissions of methane, ammonia or nitrous oxide) would contribute to such effects. Any emissions however, are likely to be offset by emissions avoided from power production by using the biogas generated.

Appendix 1			Appendix 2		Appendix 3		Appendix 4				
- Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

White et al (1995) suggests that around 193kg of carbon dioxide are produced from each tonne of waste input.

Comment on data quality:

Data given in Smith et al (2002) rely on a large number of assumptions and are not very clear. Other references are qualitative.

Potential effects:

The use of anaerobic digestion is likely to assist in controlling greenhouse emissions.

Acid gases

Number of relevant documents: 0

Summary of findings:

No information.

Comment on data quality:

No information.

Potential effects:

The anaerobic digestion process is not associated with significant acid gas emissions (although NOx may be released from biogas combustion (see Air Quality section above)) and is therefore unlikely to affect building structures. NOx emissions from associated traffic, however, may contribute to atmospheric acid gas concentrations on the local scale.

Other Potential Environmental Effects

Anaerobic digestion may be associated with traffic and visual impacts.

Appendix 1			Appendix 2		Appendix 3		Appendix 4				
Introduction	Environmental Statements	Unsegregated Incineration with Energy Recovery	Small Scale Incineration of Pre- sorted Wastes with Energy Recovery	Gasification/ Pyrolysis with Energy Recovery	Landfill with Landfill Gas Flaring and/or Energy Recovery	Composting	Mech- anical Biological Treatment (MBT)	Materials Recycling Facilities	Waste Transportation	Anaerobic Digestion with Energy Recovery	Soil Acidification

A2.12 Soil Acidification

Soil acidification occurs by means of deposition to soil of acidic gases or acid precursors released into the air from waste management processes. The effects of soil acidification include damaging trees and plants by way of stripping important nutrients from soils; disturbing the existing numbers and balance of microorganisms; reducing plant growth and therefore increase plant susceptibility to damage caused by disease, cold and high winds. The potential consequences of these effects is a loss of sensitive plant species and therefore the loss of dependant animal species, and a reduction in biodiversity; the reduction in productivity of commercial forests and farmland; the contamination of groundwater and surface water with acid and materials stripped from the soils; loss of animal and plant diversity in effected waters.

The materials which potentially contribute to soil acidification which are associated with waste management facilities include, but are not limited to:

Nitrogen dioxide- precursor of nitric acid;

Carbon dioxide- precursor of carbonic acid;

Sulphur dioxide- precursor of sulphuric acid;

Ammonia- precursor of nitric acid;

Fluoride- precursor of hydrogen fluoride;

Chloride- precursor of hydrochloric acid.

These materials are released during combustion processes (i.e. incineration, combustion of landfill gas), biodegradation of putrescible materials (i.e. composting or landfill), and movement of materials (i.e. vehicular activity).

A given habitat is capable of receiving a certain quantity of acid deposition without incurring significant levels of damage to the flora, fauna or soils. The level at which significant damage will occur is referred to as the critical load level. The critical load level is habitat specific and depends upon several factors including: the soil type, the nature of the underlying bedrock, drainage and the sensitive species type. Consequently different types of sensitive habitats will have different critical loads. In the UK critical loads for deposition of total nitrogen and total sulphur have been developed for generic habitat types.

The net potential for soil acidification associated with emissions from UK waste management operations has not been assessed. The overall contribution is likely to be small when compared to releases of ammonia from agricultural activity, releases of nitrogen dioxide, carbon dioxide and sulphur dioxide from large combustion sources (i.e. fossil fuel fired electricity generating facilities and large industrial plant) and nitrogen dioxide and carbon dioxide releases from vehicle sources. However there is the potential for elevated local soil acidification due to releases from waste management operations, which have not been quantified.

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

APPENDIX 3 : ELECTRICITY GENERATED FROM MSW ENERGY RECOVERY

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

A3.1 Introduction

The purpose of this document is to provide a quantitative summary of the usable energy generated from various municipal solid waste (MSW) management options. The MSW management options analysed include:

- Combustion (incineration) of MSW
- Combustion (incineration) of Refused Derived Fuel
- Gasification
- Pyrolysis
- Landfill disposal with landfill gas utilisation
- Anaerobic digestion with bio-gas utilisation
- Mechanical biological treatment (MBT)

In addition to the quantitative summary provided at the end of this document, a discussion of modelling assumptions and variable uncertainties is provided.

The information in this chapter can be used to evaluate the benefits which may accrue from the generation of electricity from MSW. This is an important aspect of the life-cycle assessment of waste management options, because the electricity generated from some operations will under some circumstances offset a requirement to generate electricity using fossil fuels. Avoiding the use of non-renewable energy sources will be beneficial. An initial estimate of the reduction in emissions to air resulting from the avoidance of other sources of energy is provided in this chapter. This estimate is based on reported emissions from UK energy generation in 2002. More specific estimates of emissions offsets would need to be made for consideration of individual energy use scenarios.

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

A3.2 Approach to Modelling Usable Energy and Avoided Emissions

Calculating energy yields from the waste management options requires a 'like for like' comparison of efficiency of the various technologies in converting a raw material to usable energy. Some MSW management options considered here are not energy conversion technologies, but rather methods of intermediate waste treatment. While the energy yields of some processes (e.g. incineration of MSW) are relatively straightforward calculations involving fuel input and electricity and/or heat outputs, other MSW management options (e.g. landfill gas utilisation) are characterised by greater levels of complexity and uncertainty.

Figure 1 provides a simplified overview of the interchangeable elements of energy recovery from waste processes. Quantification of usable energy from MSW management options can be sensitive to assumptions about not only about the technology in question, but other upstream or downstream processes that may be employed in the scheme.



Figure A3.1 Energy recovery from waste processes

While the interactions between various MSW management options and related upstream / downstream process can add complexity to energy calculations, these factors can be isolated. In the following sections, we highlight key modelling assumptions concerning the presumed physical characteristics of the waste stream and the expected thermal efficiency of energy conversion processes used for the recovery of usable energy from waste.

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

Avoided emissions

The emissions to air associated with a given delivery of energy to the National Grid have been estimated on the basis of total reported UK emissions from electricity generation in 2001, and the total amount of energy delivered to the National Grid in 2001. Information on emissions to air from electricity generation in 2001 was taken from the Digest of Environmental Statistics (available from <u>www.defra.gov.uk</u>) and the National Atmospheric Emissions Inventory (<u>www.naei.org.uk</u>). Information on UK electricity generation was taken from the Digest of UK Energy Statistics, 2002, available from <u>www.dti.gov.uk</u>. Dividing the UK national emissions (in grams per year) by the total electricity delivered (in kWh per year) gave the estimated emissions per unit of energy delivered (in grams per kWh). The values obtained are set out below:

Oxides of nitrogen	0.96 g/kWh
Sulphur dioxide	1.88 g/kWh
Particulate matter	0.14 g/kWh
Volatile organic compounds	0.023 g/kWh
Benzene	0.00048 g/kWh
Hydrogen chloride	0.00020 g/kWh
Hydrogen fluoride	1.4× 10 ⁻⁷ g/kWh
Cadmium	1.3× 10 ⁻⁶ g/kWh
Nickel	2.7× 10 ⁻⁵ g/kWh
Arsenic	7.1× 10 ⁻⁶ g/kWh
Mercury	4.0× 10 ⁻⁶ g/kWh
Dioxins and furans	5.1× 10 ⁻¹¹ g/kWh

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

A3.3 Characteristics of the waste stream and operations

Mixed waste streams are by definition heterogeneous and assumptions regarding the average composition of MSW must recognise variability in calorific value arising from waste pre-treatment and the processing of waste solids into gaseous fuels.

The calorific value (CV) of MSW is influenced by the extraction of recyclable and/or low CV materials. Studies by the UK Government have showed that the CV of MSW can range from between 9-11 GJ/tonne with respect to various recycling scenarios. For the purposes of this study, a recent government estimate of 9.5 GJ/tonne (GJ/Mg) is used as the base case value for MSW fuels¹.

In the case of conversion of MSW to refused derived fuels (RDF), the change in CV is more dramatic as the manufacture of RDF has the specific intent of increasing CV for more efficient energy recovery. While RDF has a higher CV (18 GJ/Mg) than MSW (9.5 GJ/Mg), there is actually little or no change in the total energy yield from RDF versus MSW on a per weight basis. This is because it takes approximately two tonnes of MSW to obtain one tonne of RDF. The energy content of RDF and MSW is therefore very similar, per tonne of raw material.ⁱⁱ

The second major consideration in evaluating the energy potential of a MSW waste stream is whether it is utilised as a solid fuel (as discussed above) or converted to a gaseous fuel through active or passive processes. Direct comparisons can be made between different energy conversion processes, but it is important to note the additional steps required to calculate usable energy when the solid waste feedstock gas been converted to a methane-rich gas feedstock.

Solid waste conversion to gas by landfilling

The rate at which solid MSW is converted to methane gas in a landfill is conditional upon several site specific factors including the size of the landfill, waste site geometry, percentage of inert materials in the waste stream, and water balances within the pit. In 1999, the Department of the Environment, Transport and Regions (DETR) issued a report investigating methane yields from landfilled wasteⁱⁱⁱ. The review of studies by government and academic researchers describes considerable variability in yield determinations with estimates ranging from 34 -146 cubic meters of methane per tonne. This wide set of values is in part attributable to varying approaches (e.g. predictive vs. empirical) to these estimations.

For the base case assumption, we have used the estimation of gas yields derived in Chapter 3 of a 1996 publication by ETSU and DTI^{iv}. The values of 200 cubic meters of gas per tonne of waste (m3/Mg) and 50% methane content (equivalent to 100 m3/Mg methane) are consistent with the ETSU / DTI report and are well cited within industry. The estimate of availability is corrected for gas losses that are expected to occur from uncontrolled leakage and migration of landfill gas beyond the site boundary.

Finally, with respect to landfill gas calculations, we have also corrected for the amount of available gas that is actually used to generate power in an average utilisation scheme. The production of landfill gas (LFG) at a typical site is not constant as the rate of production tends to increase rapidly, peak and then decay over time. Due to the high capital investment required for LFG energy recovery, power generation schemes typically utilise LFG based upon what the project developer believes will be the minimum constant yield over some time period (e.g. 20 years) rather than the total available LFG resource. For our calculations we

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

have used 50% as the net amount of gas that can be recovered for power generation following the "conservative" assumption put forward in the ETSU / DTI report.

Solid waste conversion to gas by anaerobic digestion

The production of biogas by anaerobic digestion can be designed around several biodegradable feedstocks. The technology is not suited for unsorted MSW. A direct comparison between anaerobic digestion and other MSW management options is therefore of limited value as MSW will need significant treatment before it could be considered an acceptable feedstock input for this technology. It is also important to recognise that anaerobic digestion technology is not optimised around the production of energy. In fact, a significant amount of the calorific value of the input waste stream is not converted to gas but rather remains in other outputs from the system such as compost.

The figures used to calculate usable energy from anaerobic digestion have been sourced from the Strategy Unit report, Delivering the Landfill Directive: The Role of New and Emerging Technologies and are based on a food waste stream with very high moisture content. As a result, the calorific value of this waste input may be quite different from that of MSW.

An estimate of 90 cubic metres of methane per tonne of waste has been used in our calculation of energy yield from anaerobic digestion.

Solid waste conversion to gas by pyrolysis / gasification

While gasification and pyrolysis are becoming more common in Europe, there are as yet a small number of plants in commercial operation and the availability of data on usable energy recovered from these processes is limited. To calculate the energy yield from advanced thermal treatment processes, we have taken data from a reference plant in Burgau, Germany operated by UK company WasteGen Ltd^v. While the determination of the calorific value of the gas produced by advanced thermal treatment is subject to the uncertainties regarding MSW composition and feedstock that are common to all energy recovery technologies, the CV determination is further complicated for advanced thermal treatment by the requirement for front-end sorting and separation of the MSW stream. This processing both reduces the weight of total fuel input and increases its calorific value.

The assumptions we have used in our model are based on a 200,000 tonnes per annum MSW facility with approximately 118,000 tonnes of processed waste used in energy recovery. Our estimation of energy yield is based on the tonnage of MSW received to the facility, not the processed fuel input.

To calculate energy yield on a per tonne basis, we have used WasteGen's estimation of net electricity generation, rather than gross generation. The net generation figure reflects the fact that as much as 20% of total electricity generated is consumed within the energy recovery process and is therefore not available for use.

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

Energy conversion efficiency

The amount of energy recovered in MSW management schemes is very sensitive to the efficiency of the process; that is, the rate at which heat energy contained in fuel is converted into usable energy. The two key factors influencing process efficiency are: 1) electrical efficiency of the power generation technology and 2) the amount of heat recovery. While technology choice (i.e. incineration vs. gasification) is an important determinant of process efficiency, the degree to which heat and electricity generated from the energy conversion process are utilised productively is the overriding factor.

A power plant can use the high-temperature steam produced from power generation to serve a specific heat load, either within the power generation process or outside the station gate. All of the MSW processing methods described in this report have the potential to deliver both heat and power to energy end-users but in practice, very few are configured in combined heat and power (CHP) applications. For example, of the 12 incinerators operating in the UK, only 4 of these sites are exporting heat for use beyond the plant boundary1. The productive use of heat may, in many cases, be beyond the process operator's control; factors such as the size of the scheme and local demand for heat often determine the amount of heat recovery employed in the engineering design.

When comparing the amount of energy recovered in MSW management schemes, the overall efficiency of the energy conversion process is most influenced by the extent to which electricity and heat outputs from the process are being utilised. Estimations of usable energy are often described in terms of megawatts (MW), although this can be confusing as energy output can be expressed in terms of megawatts electrical (MWe), which is the electricity-only output, or megawatts thermal (MWth), which is the heat-only output. A simple reference to MW generated usually refers to MWe.

Table A3.1 provides a comparison of the gas combustion technologies typically employed in landfill gas or biogas utilisation schemes. The figures for total system efficiency are the sums of MWe and MWth generated by each technology. Table A3.1 illustrates the large loss in efficiency that occurs when usable energy is considered to be electricity generation only. Despite the potential gains in thermal efficiency from heat recovery, productive use of heat is most frequently observed at incineration plants. MSW management options that involve transformation of solid waste to methane gas (particularly LFG utilisation) may not employ productive heat recovery due to the relatively small size of the energy recovery schemes.

¹ Nottingham and Sheffield incinerators hold contracts with district heating schemes. Coventry and Solihull hold contracts for industrial space heating.

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

Table A3.1Comparison of gas combustion technologies

Size (kW)	Electrical efficiency	Total system efficiency	Equipment type	Manufacturer name
30	26%	85%	Microturbine	Capstone
70	28%	74%	Microturbine	Ingersoll-Rand
220	35%	88%	Internal Combustion Engine	Jenbacher
1000	36%	92%	Internal Combustion Engine	Jenbacher
1750	39%	85%	Internal Combustion Engine	Cummins

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

As processes using MSW feedstocks are commonly designed as electricity-only applications, the production of heat should be assumed to be non-utilised energy, unless stated otherwise. Our comparison of usable energy from MSW management options follows this convention by providing electricity-only estimates.

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

A3.4 Results

A summary of usable energy yields from MSW waste management options is shown in Table A3.2. To facilitate a direct comparison between technologies we have made slight alterations to the categories described in the Introduction. First, we have removed two MSW management options from direct comparison with other technologies. Both mechanical biological treatment (MBT) and the manufacturing of refuse derived fuels (RDF) are processes for promoting higher thermal efficiency in energy recovery by raising the calorific value of the fuel input. As such, these processes are not energy conversion technologies per se but rather pre-treatment methods. For this reason, they have been excluded from the analysis².

Second, we have not made an effort to distinguish between gasification and pyrolysis as energy recovery technologies. With regards to advanced thermal treatment of MSW feedstocks, the distinctions between pyrolysis and gasification processes are not relevant for a straightforward comparison with other MSW management options. Indeed, many of the commercial technologies employ both pyrolysis and gasification in the thermal treatment process.

Third, we have estimated energy yield based only on electrical output. While many waste-to-energy schemes in the UK are making productive use of generated heat, there is very little public data to support generalised estimations. Usable heat energy is most commonly observed in large waste combustion facilities, but again the rate of use is highly site-specific. Heat recovery has therefore been ignored in these side-by-side comparisons.

For MSW management options involving combustion of gas, electrical conversion is assumed to occur in a gas engine with 35% electrical conversion efficiency. Solid waste incineration is assumed to occur at 22% electrical conversion efficiency.

Table A3.2 also sets out emissions from power stations which would be avoided by the generation of electricity in waste management facilities. The data in Table A3.2 does not include emissions from the waste management facilities themselves. This information is incorporated in Tables A3.3 and A3.4, which set out the net estimated emissions associated with waste management facilities. The net emissions are given by the emissions associated with the waste management facilities minus the avoided emissions from power generation. In Table A3.3, this information is given per tonne of waste processed. In Table A3.4, this information is given per kilowatt-hour of electricity generated.

² As discussed previously, pre-treatment may in any case have a negligible effect on total energy yields as increase in energy content is offset by decrease in total weight

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

	Anaerobic digestion	Pyrolysis/ gasification	Incineration	Landfill (where electricity generated)
Electricity yield	331 kWh/T	642 kWh/T	581 kWh/T	203 kWh/T
	Estimate	d avoided emissio	n (g/tonne)	
Oxides of nitrogen	318	616	557	195
Sulphur dioxide	622	1207	1093	382
Particulate matter	46	89	81	28
Volatile organic compounds	8	15	13	5
Benzene	0.16	0.31	0.28	0.10
Hydrogen chloride	0.07	0.13	0.12	0.04
Hydrogen fluoride	0.00005	0.00009	0.00008	0.00003
Cadmium	0.00042	0.00081	0.00074	0.00026
Nickel	0.0090	0.0174	0.0157	0.0055
Arsenic	0.0023	0.0045	0.0041	0.0014
Mercury	0.0013	0.0026	0.0024	0.0008
Dioxins and furans	1.7 × 10 ⁻⁸	3.2 × 10 ⁻⁸	2.9 × 10 ⁻⁸	1.0 × 10 ⁻⁸

Table A3.2 Electricity yields and avoided emissions from MSW management options

Note: The net emissions in this table are emissions from power stations which would be avoided by the generation of electricity in waste management facilities. The data in this table does not include emissions from the waste management facilities themselves (see Tables A3.3 and A3.4)

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

Table A3.3	Net emissions from MSW management options per tonne of waste processed
	net emissions nem men management options per terme er waste processea

	Net (Emission fror	emission (g/tonr n waste managen electricity	ne of waste proces nent facility minus generation)	sed) emission from
	Anaerobic digestion	Pyrolysis/ gasification	Incineration	Landfill (where electricity generated)
Oxides of nitrogen	-130 M(8)	164 M(8)	1043 G(9)	485 M(6)
Sulphur dioxide	<-619 M(8)	-1155 M(8)	-1051 G(9)	-329 M(6)
Particulate matter	No data	-77 M(8)	-43 G(9)	-23 M(6)
Volatile organic compounds	No data	-3.6 M(8)	-5.2 M(8)	1.8 M(6)
Benzene	No data	No data	No data	-0.10 M(6)
Hydrogen chloride	<-0.05 M(8)	32 M(8)	58 G(9)	3 M(6)
Hydrogen fluoride	<0.00695 M(8)	0.34 M(8)	1.0 G(9)	3.0 M(6)
Cadmium	<-0.00032 M(8)	0.068 M(8)	0.0043 G(9)	-0.00020 M(6)
Nickel	<-0.0087 M(8)	0.023 M(8)	0.034 M(8)	0.0040 M(6)
Arsenic	<-0.0018 M(8)	0.055 M(8)	0.0009 M(8)	-0.0002 M(6)
Mercury	<-0.0007 M(8)	0.066 M(8)	0.048 M(8)	0.0004 M(6)
Dioxins and furans	No data	1.6 × 10 ⁻⁸ M(8)	3.7 × 10 ⁻⁷ G(9)	1.3 × 10 ⁻⁷ M(6)

Notes:

The avoided emissions in this table are calculated from the emissions associated with the waste management facilities (from Table 6.1) minus the estimated avoided emissions from power generation (from Table A3.2).

A negative value indicates that the estimated emission from the waste management facility is less than the estimated emission from electricity generation.

A positive value indicates that the estimated emission from the waste management facility is greater than the estimated emission from electricity generation.

Data Pedigree:P(1-4): Poor; M(5-8): Moderate; G(9-12): Good; VG(13-16): Very Good

Appendix 1	Appendix 2	Appendix 3	Appendix 4
Introduction	Approaches to Modelling Usable Energy from MSW Management Options	Characteristics of the Waste Stream and Operations	Results

Table A3.4	Net	emissions	from	MSW	management	options	per	kilowatt-hour	of	electricity
generated										

	Net e (Emission fror	emission (g/kWh n waste managen electricity	of electricity gener nent facility minus generation)	rated) emission from
	Anaerobic digestion	Pyrolysis/ gasification	Incineration	Landfill (where electricity generated)
Oxides of nitrogen	-0.39 M(6)	0.26 M(6)	1.79 M(7)	2.39 P(4)
Sulphur dioxide	<-1.87 M(6)	-1.80 M(6)	-1.81 M(7)	-1.62 P(4)
Particulate matter	No data	-0.12 M(6)	-0.074 M(7)	-0.113 P(4)
Volatile organic compounds	No data	-0.0056 M(6)	-0.0090 M(6)	0.0087 P(4)
Benzene	No data	No data	No data	-0.00048 P(4)
Hydrogen chloride	<-0.00014 M(6)	0.050 M(6)	0.100 M(7)	0.015 P(4)
Hydrogen fluoride	<0.000021 M(6)	0.00053 M(6)	0.0017 M(7)	0.015 P(4)
Cadmium	<-9.6 × 10 ⁻⁷ M(6)	1.1 × 10 ⁻⁴ M(6)	$7.3 \times 10^{-6} M(7)$	-9.7 × 10 ⁻⁷ P(4)
Nickel	<-2.6 × 10 ⁻⁵ M(6)	3.5×10^{-5} M(6)	5.9×10 ⁻⁵ M(6)	$2.0 \times 10^{-5} P(4)$
Arsenic	<-5.6 × 10 ⁻⁶ M(6)	8.6 × 10 ⁻⁵ M(6)	1.5 × 10 ⁻⁶ M(6)	-1.2 × 10 ⁻⁶ P(4)
Mercury	<-2.2 × 10 ⁻⁶ M(6)	1.0×10 ⁻⁴ M(6)	8.2×10 ⁻⁵ M(6)	1.9 × 10 ⁻⁶ P(4)
Dioxins and furans	No data	2.4 × 10 ⁻¹¹ M(6)	$6.4 \times 10^{-10} \text{ M(7)}$	$6.4 \times 10^{-10} P(4)$

Notes:

The avoided emissions in this table are calculated from the emissions associated with the waste management facilities (from Table 6.1) minus the estimated avoided emissions from power generation (from Table A3.2).

A negative value indicates that the estimated emission from the waste management facility is less than the estimated emission from electricity generation.

A positive value indicates that the estimated emission from the waste management facility is greater than the estimated emission from electricity generation.

Data Pedigree:P(1-4): Poor; M(5-8): Moderate; G(9-12): Good; VG(13-16): Very Good

i Digest of UK Energy Statistics, DTI, 2002

ii Delivering the Landfill Directive: The Role of New and Emerging Technologies, Report for the Government's Strategy Unit (formerly the Cabinet Office Performance and Innovation Unit), 2002

iii Methane Emissions from UK Landfills, AEA Technology for DETR, 1999

iv Landfill Gas Development Guidelines, ETSU for the DTI, 1996

v www.wastegen.com/template.htm

vi Results from CHP Bid Pool, State of California 2001

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Statement

APPENDIX 4 : ROYAL SOCIETY'S PEER REVIEW OF DEFRA'S REPORT ON THE ENVIRONMENTAL AND HEALTH EFFECTS OF WASTE MANAGEMENT

Appendix 2

Appendix 4

Royal Society's peer review of DEFRA's report on the environmental and health effects of waste management

The Royal Society reviewed this report on the environmental and health effects of waste management and an earlier version. Both reviews can be found below. In response to substantial concerns raised in our review of the previous version of the report in November 2003 the authors presented a revised version. Our review of the final version of this report was in March 2004.



Royal Society's peer review of DEFRA's report on the environmental and health effects of waste management

November 2003

Summary

This study addresses important issues that have significant implications for the complex subject of waste policy. The authors of the report and our review group have done a considerable amount of work to bring this information together and produce this report in the short time available. In our view, the research has been comprehensive and the results are presented objectively. The collation of the data in this report is valuable but as detailed below will require further analysis before it can become the basis for making policy.

We believe that the report has significant limitations that restrict its usefulness to those making policy decisions. These limitations concern what is missing from the report and, in some places, the use made of the data reported. In particular:

- A lack of life cycle framing in this study means that any comparison of the waste management options is incomplete. No consideration is made in the results of the effects of activities displaced by recovering materials or from generating energy from the waste.
- In view of the large uncertainties associated with some of the data examined, particularly in the epidemiological studies, it would have been more appropriate to adopt a cautious approach, rather than use inadequate data in a quantitative framework. The latter may give a misleading impression of the robustness of the results.
- Caveats associated with the uncertainties in the results are not presented adequately, particularly in the quantification of the health effects, which could mislead the reader.
- The report's relevance to waste management decision-making by Local Authorities is limited, as several important issues are not addressed. These include the effect of local environmental and health sensitivity to pollutants and the impact on emissions of specific waste management activities operating under non-standard conditions.
- Bias in the availability of good quality information means the report concentrates mainly on the effects of air pollution. Consideration of the potential effects of exposure to pollutants through other pathways is not consistent throughout the report and therefore prevents adequate comparison of the options.
- Little discussion is presented as to how technological, legislative and scientific advances have affected, are affecting and will affect the management of waste in the future.

Given our concerns listed here and discussed in detail below, we recommend that this report should only be used for information and in conjunction with other reports and decision making tools that adopt a life cycle approach, such as the Environment Agency's software package WISARD (Waste integrated systems for recovery and disposal).

The Royal Society's involvement

In response to an approach from DEFRA to provide an independent peer review of this report, the Royal Society put together a working group comprising Professor Richard Perham FRS, Professor Nigel Bell, Professor Roland Clift OBE FREng, Professor Peter Guthrie OBE FREng, Professor Virginia Murray, Professor Lewis Roberts CBE FRS and Dr Lesley Rushton. The group were asked to comment on the report's comprehensiveness, familiarity with new science, objectivity and general robustness. We were not consulted regarding the study's terms of reference. This document has been approved on behalf of the Royal Society Council by Professor Sir John Enderby CBE FRS, Vice President and Physical Secretary. We welcome the fact that the Department for Environment and Rural Affairs is opening up the science information it receives to independent peer review.

Readership of the report

We recognise that this report is potentially a fundamental piece of work that has important implications for waste policy. The subject matter is complex and is hindered, as the authors recognise, by the lack of good quality studies. It is vital that the issues are addressed properly. The authors of the report suggest that it will be suitable for supporting waste management decisions at both a local and national level. However several omissions in the report (detailed below) mean it offers incomplete guidance to those making policy decisions on waste management strategies and is potentially misleading both for national policy and for local authorities. With regard to local concerns, and in addition to the misgivings detailed below, the report contains little discussion of the effect of emissions under non-standard conditions, which may be different from national averages, but are of vital local concern. Nor does it discuss the effect of local health and environmental sensitivity to the emissions.

We therefore recommend that this report should only be used for general information and be read alongside other relevant reports that take a broader life cycle approach, and which include the benefits that the various management options could provide, for example by offsetting emissions from other sources. The information in the report would be particularly useful to the Environment Agency's software package WISARD (Waste integrated systems for recovery and disposal), which enables life cycle evaluation of integrated waste management systems and is currently being updated, as the authors of the report themselves say.

Framing of the report

A major limitation to the report is the lack of a life-cycle approach to the various waste management options. We are surprised that the report has not been framed in the context of sustainable development using a life cycle approach when the *Waste Strategy 2000 for England and Wales* (DEFRA 2000) recognises the importance of such an approach in finding an overall, optimal, environmental solution for managing waste, without the risk that a decision will result in a worsening of the overall impact. Had the terms of reference, which were established before our involvement in the project, taken a life cycle approach, this report would have produced different results and allowed a better comparison of the options.

Without considering the wider issues of material and energy flows that a life-cycle approach would include, it is not possible to weigh up the full environmental and health impacts of the waste management options. For example, in Chapter 5 the results could be very different if the analysis had considered the particulates and arsenic emissions offset by incinerating Municipal Solid Waste (MSW) to generate electricity instead of burning coal. Similarly, restricting recycling to only the activities within the Material Recycling Facilities (MRF) fails to include the emissions from the reprocessing of the recyclable material and from transporting it to the recycling plant, which can be a considerable distance from the MRF. Excluding these wider considerations, and their implications on the environmental and health impacts, could lead the reader to a very skewed conclusion.

Bias of available information to air pollution

The report is mainly based on the health and environmental impacts of emissions to air, which might give the impression that impacts from alternative pathways, such as water, soil and food, are small when in fact there is a lack of good quality information. The authors recognise this and recommend several new studies to address this deficit. It is crucial that these impacts are considered if an adequate comparison of the options is to be carried out and we feel that there is not enough discussion, throughout the report, of the potential health and environmental effects.

Uncertainties

The report is inconsistent in how it presents the many uncertainties inherent in the emission data and in the use of epidemiology. In several key areas, particularly with regard to the epidemiology, the uncertainties are frequently represented inadequately and appropriate caveats regarding their assumptions and limitations are lacking or not carried through to the rest of the report. The overall effect is that the report gives an apparently reassuring estimate of the impact of different waste management options, when in fact it does not present a complete or sufficiently critical summary of the evidence. In view of the large uncertainties

associated with some of the data examined, it would have been more appropriate to adopt a cautious approach, rather than use inadequate data in a quantitative framework. These concerns are addressed in more detail below under Health impacts.

The effects on the data of using studies that include industrial, commercial and MSW, which contain substantially different quantities of degradable waste, are not discussed. For example, as the report acknowledges, MSW tends to be landfilled with other wastes including construction, demolition, commercial and industrial wastes, and sometimes with dredged material. As a significant part of these other wastes is inert and will not degrade, their presence may distort and potentially underplay the effect of MSW in these studies. This is important because if the true impacts were significant, MSW could conceivably be segregated, generating much smaller quantities that could then be dealt with using more specialist and effective techniques. Without consideration of these uncertainties and limitations in the data, the report fails to present a convincing summary of the evidence.

Although the report recognises that emissions from landfills will continue over a considerable time period and require long-term management, it makes little recognition of the changes in composition of the emitted landfill gases over time. The report does not make it clear how these changes are represented in the data or in the analysis.

Health impacts

We are particularly concerned about how uncertainties have been expressed in the quantification of the health impacts. The uncertainties in the data have been inadequately expressed in the results of the quantification and, more worryingly, data have been extrapolated to quantify the health impacts when the uncertainties demonstrate that this is inappropriate.

In particular we are concerned that in Chapter 4 the authors have extrapolated the results of the Elliott 2001 study to quantify the health outcomes when the Department of Health's Committee on Toxicity (COT 2001) concluded that it was inappropriate to draw firm conclusions on the health effects of landfill sites from this study, and that the results merited further investigation. Given the fact that the authors of this report reference the COT review of the Elliott 2001 research, we are surprised that they do not include COT's key concern that, because a study of this kind assumes that the population being measured is exposed to emissions from the landfill sites, it cannot demonstrate that the effects might be caused by other factors. Low and very low birth weights, in particular, could be related to inequalities or ethnicity, factors that have not been considered. In Chapter 3, the report provides several caveats regarding the quality of the data that Elliott uses, although the caveats are not comprehensive. For example, they do not mention that the congenital malformation register is recognised as being incomplete. Given that the authors are aware that a causal link to landfill has not been demonstrated, it would have been better if they had not attempted the notional extrapolation to produce a national figure as it could be misinterpreted. In addition the caveats are not prominent where the quantification is undertaken in Chapter 4.

The report includes the results of a comparative modelling study of the health effects of emissions from different waste technologies, using a methodology that had been developed for incineration. The health impacts are calculated using dose-response coefficients derived from the work of COMEAP. (Committee on the Medical Effects of Air Pollutants). We are concerned that the uncertainties inherent in the data in Chapter 3 and in the methodology are inadequately expressed in the results and graphs in chapter 4 and in the authors' conclusions in Chapter 7. The authors ascribe an uncertainty factor of 30 to their estimates, but attempting to compare different options when the uncertainties are so large may be misleading. In addition the results include no consideration of the effects of activities displaced by recovering materials and/or energy from the waste. This makes Figures 4.1 to 4.6 particularly susceptible to misinterpretation, as the net effects in some cases will be reduced, while in others the uncertainties will range from negative to positive incremental effects. Appropriate caveats are particularly important when dealing with sensitive issues such as deaths brought forward. We also have reservations about whether the COMEAP methodology is applicable for this kind of analysis. The limitations to this methodology are expressed in COMEAP's own report in 1998 (COMEAP 1998).

The discussion of the epidemiological evidence in Chapter 3 (3.2.1) is also limited. Confounding factors and cancer latency are important but full comprehension of the potential health effects of the different options for waste management requires

discussion of the susceptibility of populations to a particular health outcome and sensitivity to certain emissions, cumulative effects, timelines for exposure, effect of mixtures and synergies of emissions and the additive effects, for example, when combined with other environmental and occupational exposures. The latter is particularly important for workers involved in composting and material recycling facilities. Without consideration of these factors the report fails to recognise the limitations in the data.

Environmental Impact

The report makes a reasonable assessment of the uncertainties in the emission data and in the environmental effects. Recognition is made of the lack of information although the limited quantification of the impacts in Chapter 5 could give the impression that, in comparison with health, the impacts on the environment are small. Overall this may be true but as with the health impacts no mention is made of the synergistic and cumulative effects of emissions and the sensitivity of local areas for example Sites of Special Scientific Interest (SSSI). These are important considerations when considering the siting of a waste management facility.

Comparison of impacts and interpretation of the data are also made harder as the methodology is not always consistent and inclusive. For example, offset burdens of incineration and energy recovery are included only in consideration of impact on climate and not on air quality. Similarly, transport is included inconsistently and generally refers to movements of Heavy Goods Vehicles (HGV) with no reference to the movement of waste by rail and boat.

Future information needs

The report concentrates on current and historical waste management practices, at a time when the industry is changing. As the report acknowledges, changes in the legislation will mean a considerable increase in the amount of recycling and composting; however, those most affected by emissions from these processes are likely to be local residents as well as the workforce. The latter do not appear to be adequately considered in the report. Technological changes will also affect the waste management industry and may lead to substantial changes in how current practices are regarded; for example, standards may become more stringent, as history would suggest. Improvements in landfill engineering and the segregation and handling of degradable wastes could have significant effect on the emissions produced.

Whilst we agree with the authors that more research is required, particularly to improve understanding of the causal links, we believe that all analysis of waste management must be framed in a life cycle context with more consideration of the implications of legislative, technological and scientific changes that have affected and will affect the waste management industry in the future.

We believe that there will be a continuing need to update work in this area. For example we would be pleased to see a research programme set alongside national and internationally published peer review data that recognises the need to reduce uncertainties. In our view it is essential to share this evolving information with decision makers and the public.

We have not reviewed the extended summary of this report on the environmental and health effects of waste management.

This review and our involvement in the study can also be found on the Royal Society web site: www.royalsoc.ac.uk. For further information please contact Richard Heap in the Science Advice Section, Royal Society, 6-9 Carlton House Terrace, London SW1Y 5AG. Email: science.advice@royalsoc.ac.uk.

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THE ROYAL Society

Royal Society's peer review of DEFRA's report on the environmental and health effects of waste management

March 2004

The revisions made by the authors of this report in response to the Royal Society's review of the previous version in November 2003 have been considered on behalf of the Royal Society by the chair of the Society's working group, Professor Richard Perham FRS, and by the chair of the Society's review panel and Vice-President, Professor Sir John Enderby CBE FRS. It has not been practicable to reconvene the full review panel to comment on the revisions.

The Royal Society is satisfied that a significant number of its concerns have been addressed.

Throughout our review we have stressed the need to clarify the uncertainties inherent in the data in this report and consider the implications this uncertainty has when evaluating the environmental and health effects of waste management. Although the uncertainties have been acknowledged in this report, it is important that anyone using these data takes adequate consideration of its inherent uncertainty.

In Section 1.1 the authors refer to a separate report prepared by Eftec and Enviros Consulting Limited for DEFRA on the economic costs and benefits of health and environmental effects of waste management. We have not seen or been asked to review this separate report. However we have been assured by DEFRA that it will give adequate consideration to the uncertainties inherent in the data on the health and environment effects.

We have not reviewed the extended summary of this report on the environmental and health effects of waste management.

Professor Perham and Sir John Enderby acknowledge with gratitude the support of Richard Heap, Royal Society secretariat.