INTARESE

Project No. 018385

INTARESE

Integrated Assessment of Health Risks of Environmental Stressors in Europe

Integrated Project

Thematic Priority

Deliverable reference and title

WP 3.6 Waste Assessment report

Due date of deliverable: February 2009

Actual submission date: March 6, 2009

Start Date of Project: November 2005	Duration: 42
Organisation name of lead contractor for this deliverable:	
ASL	

Project co-funded by the European Commission with the Sixth Framework Programme (2002-2006)		
Dissemination Level		
PU	Public	
PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	
CO	Confidential, only for members of the consortium (including the Commission Services)	x

WP3.6 WASTE

1ST PASS ASSESSMENT REPORT

MARCH 6, 2009

Authors

Francesco Forastiere (ASL), Chiara Badaloni (ASL), De Hoogh K (IC), Martin Krayer von Kraus (WHO), Marco Martuzzi (WHO), Francesco Mitis (WHO), Lubica Palkovicova (RB-SMU), Daniela Porta (ASL), Philipp Preiss (USTUTT), Andrea Ranzi (ARPA Emilia Romagna), Carlo A Perucci (ASL), and David Briggs (IC).

Aknowledgements

We acknowledge comments and useful criticisms from Alfred Barnard (UCL), Anastasios Karabelas (CERTH), Alexandra Kuhn (USTUTT), Tomas Trnovec (RB-SMU), and Annunziata Faustini (ASL).

We thank the following persons for the case study in Italy: Antonio Lazzarino (ASL), Simona Milani (ASL), Giulia Cesaroni (ASL), Laura Erspamer (ARPA Emilia Romagna), Paolo Lauriola (ARPA Emilia Romagna), Giuseppe Viviano (ISS), Andrea Paina (APAT), Valeria Frittelloni (APAT), Nicola Caranci (Emilia-Romagna ASSR).

We thank the following persons for the case study in Slovakia: Nada Machkova (Slovak Environmental Agency - SEA), Alexander Jancarik (SEA), Katarina Paluchova (SEA), Jana Babjakova (RB-SMU), Vladimir Svabik (OLO BA), Katarina Stebnicka (KOSIT), Jozef Pecho (SHMI), Peter Teslik (SO SR), Peter Heidinger (SO SR).

We thank Patrizia Compagnucci (ASL) and Margareth Becker for their help in editing the manuscript.

Table of contents

Abstract

We have conducted a health impact assessment of landfilling and incineration in three European countries - Italy (It), Slovakia (Sk) and England (En), - which have different waste policies. The overall aim was to develop an assessment model that could be applied to the baseline scenario in 2001, in view of evaluating a variety of waste management options in the 2nd pass assessment. For each of the three countries, national waste management policies have been described, total emissions arising from waste management have been estimated, census data on incinerators and landfill sites in the three countries was collected and population exposure was estimated. Country level information on all incinerators and landfills was collected and transferred to a Geographic Information System (GIS) system, and the populations living within 3 km of an incinerator and 2 km of a landfill were estimated. Air pollution dispersion modelling was applied to the areas where incinerators were present and particulate matter (PM) and nitrogen dioxide (NO_2) concentrations were estimated. A systematic review of the epidemiologic literature was conducted to derive the appropriate coefficients of the health impact. For incinerators, we estimated attributable cancer incidence and years of life lost (YoLL), while for landfill sites we estimated attributable cases of adverse reproductive outcomes (congenital anomalies and low-birth weight infants). We have also conducted an evaluation of the impact of incineration in terms of Years of Life Lost for the entire European population based on a large scale dispersion model. The uncertainties in the assessment were systematically evaluated and a confidence level was assigned to each step.

A total of 560, 283 and 587 kg of municipal waste per inhabitant were produced in Italy, Slovakia and England in 2001, respectively. The processing of municipal waste led to emissions to the environment of several grams of toxic substances per inhabitant, with Italy being the greatest emitter of metals (nickel and arsenic) and dioxins. In 2001, 49 (It), 2 (Sk), and 11 (En) incinerators were operating, with an estimated population of 1,000,000 (It), 16,000 (SK) and 1,200,000 (En) subjects living within 3 km of an incinerator. There was a clear association between living close to incinerators and low socioeconomic status in Italy and England whereas the opposite was found for Slovakia. The total additional contribution to PM_{10} and NO_2 within a 3 Km radius of incinerators was estimated using ADMS air dispersion modelling under an emission scenario corresponding to national limits. Based on these estimates, we have high confidence that the total annual average additional contribution to PM_{10} concentration within 3 km is 0.0114 ug/m^3 for Italy, 0.0078 ug/m^3 for Slovakia, and 0.0017 ug/m^3 for England. We have high confidence that the additional annual average contribution to NO_2 is 0.2271 ug/m^3 for Italy, 0.1542 ug/m^3 for Slovakia, and 0.1438 ug/m^3 for England. The use of measured emission values, instead of limits based on the legislation, had a strong impact on the estimate for PM_{10} (e.g. 0.0030 ug/m^3 for Italy) but a lower impact for NO_2 (e.g. 0.1944 ug/m^3 for Italy).

For the period 2001-2050, we estimated the annual number of cancer cases “attributable” to exposure before 2001 (“past exposure”) and to exposure during 2001-2020 (“current exposure”). Assuming that all the incinerators operating in 2001 continued to operate until 2020, we are moderately confident that about 90 (It), less than one (Sk), and 36 (En) cases per year will be attributable to past exposure to emissions from incinerators up to 2020 and then the number will decline to almost zero in 2050. On the other hand, the annual number of cases due to current exposure in 2001-2020 increases to 11 (It), 0.071 (SK), and 7 (En) in 2020 and then will decline to 0 in 2050. Through a life tables analysis, we were able to estimate with moderate confidence that by 2050, the attributable impact on the 2001 cohort of exposed residents will be 3603 (It), 181 (Sk) and 4217 (En) Years of Life Lost (YoLL). When the effects of emissions from incineration was evaluated for Europe using large scale dispersion models for the emissions of several pollutants, the estimated YoLL per year were 38.6 (real emission values) or 56.4 (national limits) for plants in Italy, 2.45 for plants in Slovakia, and 24.1 for plants in England.

A total of 118 landfills have been surveyed in Italy, 121 in Slovakia and 232 in England (all operating in 2001), with a total exposed population of 1,350,000 (It), 329,000 (Sk) and 1,425,000. In Italy and England, but not in Slovakia, the population living close to landfills tend to be of a lower socioeconomic status. Assuming that the emissions resulting from the pre-2001 operation of these landfills will continue until 2030, we have moderate confidence that the average annual number of additional cases of congenital anomalies and newborns with low birth weight will be 1.96 and 42.4 (It), 1.54 and 12.7 (Sk), and 2.7 and 58.5 (En), respectively.

In conclusion, there are several uncertainties and critical assumptions in the assessment model that are typical of a complex problem such as the evaluation of the health impacts of waste management policies. However, we believe that the model provides insight into the relative health impact attributable to incineration and landfilling, and that it could potentially be useful for evaluating future changes of European and national policies regarding waste prevention and minimization, recycling, landfill closures and incineration with energy production.

2. The issue

1.1 Introductory rationale

Waste is an environmental, social and economic challenge for developed societies. An average of 3.5 tonnes of waste per person per year is generated in Europe. This is mainly made up of waste coming from households, commercial activities, industry, agriculture, construction and demolition projects, mining and quarrying activities and from the generation of energy.

"Waste management", that is the generation, collection, processing, transport and disposal of waste, is important for both environmental reasons and the health of the public. Overall, the volume of waste is growing in Europe. With higher levels of economic growth anticipated, overall volume growth is predicted to continue and will concern most wastes. Municipal solid waste (MSW) generation has been contributing significantly to this growth and it is connected to the level of economic activity. Municipal solid waste represents approximately 14% of all waste produced and consists mainly of paper and cardboard (35%), organic material (25%), plastic, glass, ferrous material, textiles, aluminum, and other types of waste.

On average, each European citizen generated 460 kg municipal waste in 1995. This amount rose to 520 kg per person by 2004, and a further increase to 680 kg per person is projected by 2020. In total, this corresponds to an increase of almost 50% in 25 years. This projected continuing increase in waste volumes is primarily due to an assumed sustained growth in private final consumption and a continuation of current trends in consumption patterns (EEA, 2008).

With large quantities of waste being produced, it is important that its management causes as little harm as possible both to human health and to the environment. There are a number of different options available for the treatment and management of waste including prevention, minimisation, recycling, energy recovery and disposal (Strange, 2002; European Topic Centre on Resource and Waste Management http://waste.eionet.europa.eu/etcwmf). An increasing amount of the resources contained in waste is recovered as materials or as energy in incinerators or biogas plants, but approximately half is still permanently lost in landfills.

Waste management is one of the key priorities of EU environmental policy and the framework in this area has been progressively put in place since the 1970s. Efforts are being made to decouple waste production from economic development through a combination of waste prevention, recycling and reuse; where disposal is necessary incineration is preferred over landfill. The amended Waste Framework directive, final adoption of which is expected by 2009, will be an important step towards further coordinating efforts in this area. The Sixth Environment Action Programme (2002-2012) set the level of ambition for the further evolution of European waste management policy when it called for: a decoupling of environmental pressures from economic growth, and a significant reduction in (i) volume of waste generated, (ii) quantity of waste going to disposal (i.e. landfill or incineration with no or low rates of energy recovery), and (iii) volumes of hazardous waste produced. This level of ambition was repeated in the European Commission's proposed Thematic Strategy on Prevention and Recycling (2005): 'The long-term goal for the EU is to become a recycling society that seeks to avoid waste and uses waste as a resource' (EEA, 2008).

Concerns remain about potential health effects associated with the main waste management technologies (incineration, landfilling). Because of the wide range of pollutants that may be released, the different pathways of exposure, usually long-term low-level exposure, and a potential for synergistic and cumulative effects, there are many uncertainties involved in the assessment of health effects in populations exposed to emissions derived from waste management technologies. It has been recently underlined that "controversy and consequent difficulties in developing and adopting health–friendly, cost-effective and equitable policies in waste management are due to several reasons, including: abundance of suggestive, but not conclusive, evidence on possible adverse health effects of living near waste treatment plants, landfills, incinerators etc; confusion between different issues such as the disposal of solid urban waste as opposed to other types of waste (for example, toxic industrial waste, hospital waste); mistrust in authorities and scientific community; occurrence of "not in my backyard (Nimby) syndrome" type of reaction, possibly as a result of overlooking aspects of risk perception and communication." (WHO, 2007)

One important issue in waste management that Europe is facing is the illegal practice of waste dumping or waste burning. These practices are not accounted for in official statistics but it is known that they are present in many European places (e.g. Campania in Southern Italy see Box 1) and the environmental and health consequences have not been quantified.

This Assessment report is focused on the health effects of MSW. At present in the EU, municipal solid waste is disposed of through landfill (49%), incineration (18%), recycling and composting (33%). There are wide discrepancies between Member States, ranging from those that recycle least (90% landfill, 10% recycling and energy recovery) to those which are more environmentally friendly (10% landfill, 25% energy recovery and 65% recycling). A recent report from the European Environmental Agency (EEA, 2007) underlined the necessity to divert municipal waste from landfill, based on the concept of "waste hierarchy"; this means that prevention/reduction of waste is the most preferred and waste disposal is the least preferred options for waste management; reuse, recycling and recovery falls in between these two. The worst option represents the landfilling of untreated waste, because of its emissions of methane, its long-term emissions to soil and groundwater as well as the loss of the resources it contains. The EEA report categorized European countries into three waste management groups according to the strategies for diversion of MSW away from landfill and the relative shares of landfilling, material recovery (recycling and composting) and incineration. For instance, Italy belongs to the 2nd group, meaning countries with high material recovery and low incineration (incineration < 25%, material recovery > 25% + medium dependence on landfill) whereas England and Slovakia both belong to the 3rd group of countries with low material recovery and low incineration (incineration < 25%, material recovery < 25% + high dependence on landfill), although England is now close to joining Group 2, since in 2005 it recycled 27% of MSW (DEFRA, 2006).

1.2 The Assessment Framework

The overall aim of the present case study was to assess potential exposures and health effects arising from municipal solid wastes throughout their lifecycle, from generation to disposal or treatment. We have performed the assessment at the country level as we have initially used Italian data as an example (data from region Emilia-Romagna was used first in a pilot evaluation), followed by England and Slovakia assessments.

In our assessment model, we evaluated the health impact of different management policies for MSW considering a baseline scenario for the year 2001. For practical reasons, it was hypothesized that the baseline scenario would continue to operate with no changes until 2020, and the health impact has been estimated up to 2050. The main effort has been the development of the instruments for the evaluation and no specific policy issues are addressed in this report. We have been mainly focusing on problems in data collection, methods for calculation of emissions, modelling exposure, and finding exposure-response functions so as to perform the assessment of the current situation. The methods implemented for the baseline scenario will be a useful instrument in the 2nd pass of the project to evaluate the changes that are currently occurring and to respond to policy questions arising from future developments.

1.3 Key Elements/Relationships for Waste Assessment

The assessment protocol has been following the full chain approach illustrated in Figure 1, and the key elements of the assessment are illustrated below.

We have divided the process into the following different key elements according to the full chain approach.

From generation to management of waste. Describe waste generation and waste management policies for each country.

From waste management to emissions of pollutants. Given the baseline scenario, evaluate emission data for the main waste technologies and estimate total emissions of air pollutants with a potential health effect at the country level. The limitation of air emissions is discussed in the next paragraph.

From emissions to population exposures. Provide an estimate of the size of the population exposed and the level of exposure to pollutants emitted from the main management technologies at the country level.

From exposure to health effects. Perform a systematic review of the scientific literature and derive appropriate relative risk estimates and exposure-response functions.

Quantification of the health impact. Estimate the integrated health impact attributable to waste management at the country level.

Quantification of the external costs. Estimate external economical costs of waste management practices at the country level.

1.4 Important exclusions or assumptions

There are some specifications and key choices that are important to consider in this framework.

There are substantial environmental emissions associated with waste transport for both recycling and disposal. While we have considered transportation in the evaluation of total emissions, the health impact from transportation has not been evaluated in detail and it will be considered in the 2nd pass.

Although the phenomenon has been a matter of great concern in Italy in recent years due to the waste crisis in Campania (see box 1), the quantification of illegal practices of dumping and burning is extremely difficult and a formal health assessment was considered premature.

The emission factors that we considered are based on facilities under normal operational circumstances. There is the possibility of accidental releases that should be considered but are difficult to quantify.

Although all major waste management activities have been considered for steps 1 and 2, the focus for the additional steps are based on incinerators and landfills representing the main methods of waste disposal in the baseline scenario.

Although pollutants from waste disposal practices are released into all environments (not only air, but also water, soil), only emissions into ambient air have been taken into consideration in the full assessment, due to the lack of data on emissions into soil and water and the complex issue of appropriate exposure-response functions.

We have limited our evaluation on emissions of air pollutants with potential direct health effects and we did not consider the impact of greenhouse gases emissions. This is an area of great concern (EEA, 2008) that deserves a specific approach and a different methodology.

Cost evaluation is the last point of the evaluation and it is important for present and future scenarios. However, agreed upon methods should be developed within INTARESE and will be considered in more detail in the second pass in accordance to the standard approach utilized by all partners.

1.5 Evaluation of uncertainties

Identification of major sources of uncertainty has been considered in the assessment. We have systematically tried to state for each step of the evaluation the level of confidence using the scale proposed in the IPCC document (IPCC, 2005) A *level of confidence* can be used to characterize uncertainty based on expert judgment as to the correctness of a model, an analysis or a statement. The following scale was adopted to express the level of confidence in the correctness of the estimate : very high confidence at least 9 out of 10; high confidence about 8 out of 10; moderate confidence about 5 out of 10; low confidence about 2 out of 10; and very low confidence less than 2 out of 10.

BOX 1. Illegal management of waste and health effects

Most of the available studies on the potential health effects of landfills were carried out in reasonably controlled settings, i.e. where waste disposal is managed applying relatively tight and regulated practices, aiming at minimizing releases of toxic agents through air, soil and water contamination. Less is known of the health effects of waste-related exposures in settings with lower standards of waste management practices, such as the case of Campania, a region in southern Italy known for its problematic waste situation. The northern part of the region, consisting of two of the five provinces, Naples and Caserta, has frequently been in the news, over the last 15-20 years, because of the periodic crises in public-run waste collection services. Well known operations have also been run, since the early 1980s, by organized crime cartels, resulting in documented practices of illegal dumping and open-air burning of urban and toxic waste (Legambiente-Osservatorio Ambiente e Legalità, 2007). Waste management in the whole Campania region has been run under a central government-declared emergency regime since 1994. In short, a difficult to quantify but large proportion of waste produced in the region, plus waste transferred into the region by organized crime, has been illegally disposed of and burned for some two decades.

A major reason of concern, of increasing prominence in the often controversial public debate, has regarded the health effects of such practices. Acute effects, such as the possible outbreak of vector-borne infections and general safety issues, are of obvious concern, but possible long-term health effects also attract much attention, given the protracted emergency status. Insights from Campania may be valuable elsewhere: recent reports from international agencies indicate that waste mismanagement occurs in Europe and beyond, with substantial illegal shipment of hazardous waste, especially from Organisation for Economic Co-operation and Development (OECD) countries to new EU Member States, Balkan countries and the Commonwealth of Independent States (European Topic Centre on Resource and Waste Management, 2008) and substantial quantities of waste dumped in illegal sites (European Environmental Agency, 2008). In Slovakia, for instance, there are two kinds of illegal dumping sites: those containing communal and construction waste and stores of obsolete pesticides previously used by agricultural cooperatives during the socialist period. Illegal dumping sites containing plastic materials (e.g. PVC) occasionally start to burn producing polychlorinated dioxins and dibenzofurans as the result of incomplete combustion.

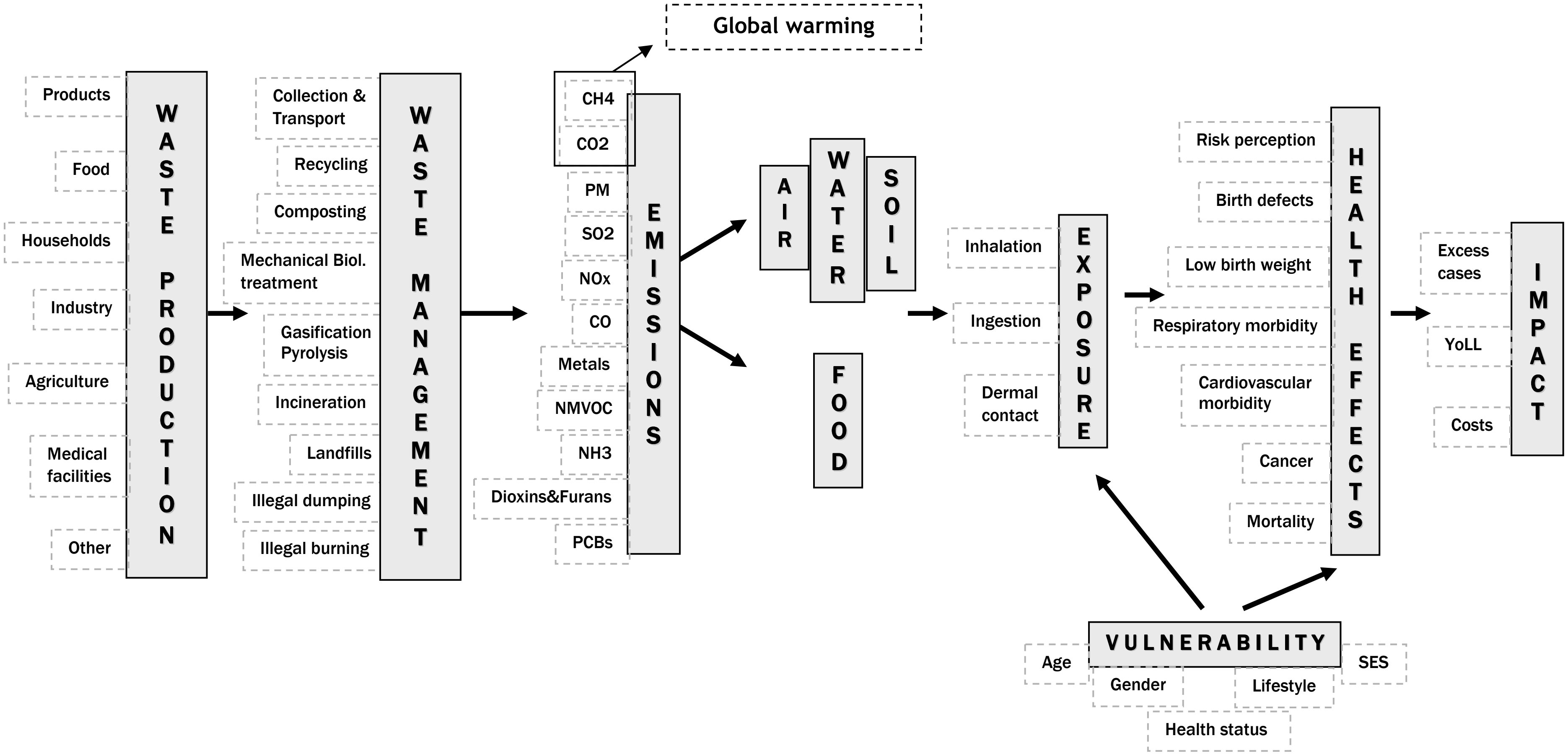
References

Legambiente-Osservatorio Ambiente e Legalità (2007). Rapporto Ecomafia 2007. I numeri e le storie della criminalità ambientale (in Italian). Rome: Legambiente.

European Environmental Agency. Sustainable consumption and production in South East Europe and Eastern Europe, Caucasus and Central Asia. Joint UNEP-EEA Report on the opportunities and lessons learned. Copenhagen, Geneve: EEA Report No 3/2007 (available at http://reports.eea.europa.eu/eea_report_2007_3/en/, accessed on April 2008).

European Topic Centre on Resource and Waste Management. Transboundary shipments of waste in the EU. Developments 1995-2005 and possible drivers. Copenhagen; 2008 (available at http://eea.eionet.europa.eu/Public/irc/eionet-circle/etc_waste/library?l=/working_papers/shipments290208pdf/_EN_1.0_&a=d, accessed on April 2008).

Figure 1: The full chain approach - from waste production to health effects



2. Assessment Methodology

2.1 Waste Generation and Management in Italy, Slovakia and England

We have specified the waste management technologies of our interest in Appendix 1 (recycling, composting, mechanical and biological treatment, anaerobic digestion, gasification/pyrolysis, incineration, landfill) whereas Appendix 2 briefly summarizes the EU legislation on the issue.

We aimed to describe the current waste management policies in the three countries. We searched data on wastes collected by the National Statistical Institutes and Ministries for the Environment for each year and reported to Eurostat every second year (*http://epp.eurostat.ec.europa.eu*). Data are separated for municipal waste and waste from industry and trade, both hazardous and non hazardous waste. The European Topic Centre on Resource and Waste Management (*http://waste.eionet.europa.eu/wastebase)* provided a data set *(*Wastebase) with detailed information of the policies at the country level. On the other hand, only partial information on location data of major landfills and incinerators in Europe are available in the EPER database (*http://www.eper.cec.eu.int/eper/default.asp*).

Country-specific sources of information have been considered, like the Environment Agency in the UK or the Slovak Environmental Agency. Data about Italian MSW production and management have been provided by the Italian agency for the protection of the environment and for technical services (APAT - *Agenzia per la Protezione dell'Ambiente e per i servizi Tecnici*). Data were checked for consistency and integrated with information collected from an independent body, the National Waste Observatory (ONR - *Osservatorio Nazionale Rifiuti*).

For each country, we have collected information on the following list of indicators (for the years 2001 or 2002 for Slovakia).

Total MSW generated (and per inhabitant)

- Number of materials recycling facilities
- Number of composting facilities
- Number of mechanical and biological treatment facilities
- Number of incineration facilities
- Number of landfills
- Amount and percentage of MSW recovered for recycling
- Amount and percentage of MSW recovered for composting
- Amount and percentage of MSW treated with mechanical and biological treatment
- Amount and percentage of MSW incinerated
- Amount and percentage of MSW in landfill disposal

2.2 Quantification of Emissions of Pollutants from Waste Management in Italy, Slovakia and England

The next step in the full-chain approach was the assessment of the emissions from waste management into the environment. A description of the main emissions from the most important waste technologies is briefly presented in Appendix 1 on the basis of information available in the UK report (Enviros, 2004).

2.2.1 Overview of Emissions to Air from Waste Management Facilities

We searched for information on emissions into the air for the processes described in the section above. We found that the most complete source is the report from DEFRA in UK (Enviros, 2004, table 2.45 page 115). This report has considered all the available literature and provides a unique source of information of estimated emissions in grams per Tonne of processed waste. The emission values have been calculated on the bases of various sources of information, the most important being the industry returns to the pollution inventory and Environment Agency/DEFRA sponsored research. Additional data were from plants operators and from already published research. The reliability of the information from which the numerical data have been derived has been assessed and quality of the information has been scored according to the specific "data pedigree". For example, a high level of reliability was assigned to direct measurements - rather than proxy-based measurements – and to those estimates that were based on a large number of field measurements, whose data have been obtained with widely used approaches and best practice, and whose results have been cross-checked. As a consequence, the quality of these estimates differs from one kind of treatment to another. For example, the data pedigree of MBT and landfill is considered "moderate", whereas it is considered "good" for incineration. The report also used information collected in the documents prepared by the National Society for Clean Air in 2002 (NSCA, 2002). In addition, a more recent reference document for the best available technique for waste incineration (http://eippcb.jrc.es/pages/FActivities.htm) provides indications on current emissions for incineration plants.

For Italian incinerators we used data collected in the Emilia-Romagna region under a long term program of evaluation of emissions from eight plants located in the area (Morselli et al. 2007, 2008). We considered detailed emission data monitored in 1996 and in 2003 and estimated the average values of the two as applicable for 2001 for our purposes (Table 1).

The pollutants (emissions into air) of interests were:

- Primary Particulate matter (PM) as PM_{10}, $PM_{2.5}$ and coarse fraction
- Cadmium
- Nickel
- Arsenic
- Mercury
- NOx
- SO2
- HCl
- HF
- Dioxins/Furans
- PCBs

2.2.2 Quantification of emissions from waste management processes

The total amount of MSW produced was divided according to its management destination. It should be specified that some treatment processes produce an amount of residual materials per amount of MWS, depending on the kind of facility. Such residuals enter again into the flow of MSW management and therefore the actual amount of MSW treated in the facilities is different from the amount of MSW initially directed to them. Data on these residuals have been collected and included in the calculations. It has been assumed that MBT (mechanical biological treatment) facilities produce about 50% of material to be incinerated and 17% of material to be landfilled; in addition, it has been assumed that the residuals to be landfilled from incinerators are about 28% of the material originally delivered. Because this data was developed under circumstances which are considered as “standard” for MBT facilities and incinerators, and considering that consistent values have been reported in the descriptive waste reports of the three countries, we have a high level of confidence in these assumptions.

We have applied the emission factors in Table 1 for incinerators and Table 2 for other processes (grams per Tonne) to estimate the process-specific and total air emissions of the pollutants at the country level. While we have a high level of confidence for the emission factors related to incinerators (as they are based on measured values for Italy and the UK), we have only moderate confidence in the values for the other technologies because they were estimated for England and extrapolated to the other countries. The results have been divided by the total population of the country to obtain emitted toxicants per inhabitant. All the calculations have been performed using an Excel spreadsheet (available upon request). The spreadsheet is a useful instrument for the second pass assessment as it allows performing calculations under different scenarios.

Table 1. Emission factors (gramms per tonnes of municipal waste) from incinerators in the three countries in 2001.

Pollutant	Emilia-Romagna *		Italy	Slovakia	England
	1996	2003	2001**	2001***	2001
PM	28	22	25	38	38
Cadmium	0.05	0.02	0.04	0.05	0.05
Nickel	1.36	0.10	0.73	0.05	0.05
Arsenic	0.026	0.040	0.033	0.005	0.005
Mercury	0.27	0.13	0.20	0.05	0.05
NO_x	1598	1290	1444	1600	1600
SO_2	128	73	101	42	42
HCl	129	31	80	58	58
HF	2.4	2.8	2.6	1	1
Dioxins/Furans	1.2E-04	3.2E-05	7.6E-05	4.0E-07	4.0E-07
PCBs	3.0E-05	3.0E-05	3.0E-05	1.0E-04	1.0E-04

* Measured values from eight plants in Emilia Romagna

** Italian emission factors for 2001 estimated as average of 1996 and 2003 data from Emilia-Romagna

*** Slovakia emission data assumed to be the same as for England

Table 2. Emission factors (gramms per tonnes of municipal waste) from management processes* in the three countries in 2001.

Pollutant	MBT	Anaer. Digest.	Pyro./gas.	Landfill/engines	Landfill/flaring	Transport
PM	0	0	12	5.3	6.1	1.3
Cadmium	0	0.0001	0.0069	0.0071	0.0071	0
Nickel	0	0.0003	0.04	0.0095	0.0095	0
Arsenic	0	0.0005	0.06	0.0012	0.0012	0
Mercury	0	0.0006	0.069	0.0012	0.0012	0
NO_x	72.3	188	780	680	75	31
SO_2	28	3	52	53	90	0.11
HCl	1.2	0.02	32	3	14	0
HF	0.4	0.007	0.34	3	2.7	0
VOCs	36	0	11	6.4	7.6	5.1
Cl-VOCs	0	0.0004	0	2.77	2.63	0
Benzene	0	0	0	0.00006	0.00006	0.0029
Dioxins/Furans	4.0E-08	0	4.8E-08	1.4E-07	5.5E-08	3.8E-11
PCBs	0	0	0	0	0	0

* Emission factors for incinerators are presented in table 1.

Adapted from Enviros (2004)

2.3 Population Exposure to Emissions from Incinerators and Landfills in Italy, Slovakia and England

The next step in the health impact assessment was to identify and characterise the population at risk near the plants and to estimate population exposure. Briefly, information on all incinerators and landfills at the country level has been collected, the information transferred to a GIS system, and the population living within a specific radius from both incinerators and landfills has been estimated. Estimates were made by an indicator of socioeconomic status at small area level (census tract level or postcode district), i.e. deprivation index. Air pollution dispersion modelling has been applied to the areas where incinerators are present. These steps are illustrated below.

2.3.1 Census and GIS coordinates of incinerators and landfills

Data for all landfills and incinerators in the three countries have been collected according to a specific protocol from national sources as described below. We could not use the European Pollutant Emission Register (EPER) data-base as it contains data only from facilities that are obligated to report their emissions to the EPER (municipal waste >3t/h).

We have used the following GIS coordinate systems; the British National Grid (BNG) for England and Wales, the Transverse_Mercator (WSG_1984_UTM_Zone_33N) for Italy and Krovak (S-JTSK_Krovak_East_North) for Slovakia.

Italy

The Italian Environmental Protection Agency (www.apat.gov.it) provided a database of the incinerators operating during the period 2001-2007. In addition, a detailed census of the 52 incinerators operating in 2005 was made by a national research institute (ENEA, 2007) funded by the national association of stakeholders in waste management (Federambiente). Detailed data were also provided by the regional environmental authority of Emilia

Romagna for all eight incinerators located in this region. From all these sources of information, we were able to single out the 40 incineration plants operating in 2001, define their GIS coordinates and get specific information on year operations began, number of lines, fumes capacity (Nmc/h), stack height (m), stack diameter (m), exit velocity (m/s),emission rate (m3/s), and exit temperature (°C). In a few cases, when the information on technical characteristics was missing, it was approximated using information from other plants with similar characteristics. These approximations, although relatively important for the single plant, were likely to have had only a minor impact on the overall assessment.

The Italian Environmental Protection Agency provided a database of the landfills in Italy (a total of 619 in 2001) with information of the total capacity and waste land filled per year. Unfortunately, the data base did not contain GIS coordinates and it was impossible to retrieve the information for the entire country. Using contacts with regional environmental authorities, we were able to get geographical information for five regions (Piemonte and Emilia Romagna (North), Toscana and Abruzzi (Centre), and Campania (South)) for a total of 118 landfills. For the rest of the country, we assumed (with a moderate level of confidence) that the characteristics (sex, age and socioeconomic status) of people around the 501 missing landfills were similar to those of the 118 studied sites.

Slovakia

Basic information on incinerator census for the baseline year 2001 together with information on the number of incinerators handling municipal waste have been obtained from the Slovak Environmental Agency (SEA) managed by Slovak Ministry of Environment. There were two incinerators for MSW in 2001 and detailed information on GIS coordinates, year operations began, number of lines, fumes capacity (Nmc/h), stack height and diameter (m), exit velocity (m/s), emission rate (m3/s), exit temperature (°C) was obtained directly from companies managing both incinerators – joint stock companies OLO in Bratislava city and Kosit in Kosice city.

At the end of 2001, there were 165 active landfills for municipal wastes in Slovakia. The list of landfills according to region was available (in Slovak) from the website of the Slovak Ministry of Environment (www.enviro.gov.sk/servlets/files/14374). Furthermore, detailed information on GIS coordinates, capacity and year of start was obtained again from the Slovak Environmental Agency. Out of 165 active landfills in 2001, 121 were geocoded. We assumed (with a high level of confidence) that the characteristics (sex, age and socioeconomic status) of people around the 45 missing landfills are similar to those of the 121 studied sites.

England

This study used all 11 municipal waste incinerators operating in England during 2001. Data on emission of numerous substances (Tonnes/annum) and location (x- and y-coordinates) for these incinerators were obtained from the Environment Agency (EA). Data necessary for conducting the atmospheric dispersion modelling (e.g. stack height, stack diameter, emission rate etc) were, if possible, sourced from the waste companies websites. Where no data could be found an average was taken from the known incinerator data and applied.

For this study, data for all regulated landfill sites in England and Wales was obtained from the EA. No data for 2001 was available because of changes in the permitting regime. In 2001 landfill sites came under a different directive and were not required to report to the EA under the Pollution Prevention and Control (PPC) Regulations. The EA advised to use the 2006 landfill data instead as a good indicator for the 2001 situation (personal communication Tom Ash, EA, October 2007). The 2006 dataset contains information about 242 regulated landfill sites (e.g. northing, easting, atmospheric releases and threshold of waste). The "real" number of landfill sites in England is considerably higher than the number received from the EA.

2.3.2 Population data by gender, age and socioeconomic status

Population data at the smallest unit of aggregation for the census 2001 were available for the census blocks in Italy (about 100-200 (mean 162, sd 223) inhabitants per unit) and Slovakia (about 700-800 (mean 785, sd 1318) inhabitants). These geocoded files were available from the National Institute of Statistics in Italy (ISTAT) and the Slovak Environment Agency. In Great Britain, the Royal Mail maintains a country-wide system of postcodes to identify postal delivery areas and allow statistics to be created based on postcode as a main geographic reference. The postcode data was extracted from Codepoint, a product from the Gridlink consortium. Postcode populations (by 5 year age bands and sex) were calculated as part of a SAHSU study, whereby census population data were disaggregated to postcode level (mean 41, sd 37 for GB). For each census block, population density (inhabitants/m^2) was calculated to estimate population size for subdivisions of the census blocks.

For each census block in Italy, a deprivation index was available from a national project funded by the Ministry of Health (Nicola Caranci, personal communication). A similar application, using 1991 data, has been published (Cadum et al. 1999). The following census information that represents various aspects of deprivation were included in the index: education, occupation, home ownership, family composition and nationality. An algebraic combination of these factors was used to create an index of socioeconomic position by census block, with the corresponding population distributed in quintiles, ranging from very well off (level 1) to very underprivileged (level 5).

For Slovakia, an index of socioeconomic position was derived from the following census variables: education (proportion of population with university, secondary, basic or no education), proportion of children, proportion of employed among 16-64 year olds, house type (house or flat), and house ownership. Again an index per census block was distributed in quintiles, ranging from very well off (level 1) to very underprivileged (level 5).

For England, the Carstairs score, which is based on four census variables (lack of car ownership, unemployed head of the household, low social class and overcrowding) was applied as the deprivation index. The Carstairs score is available at the smallest census area, the output areas (OAs). By means of a point-in-polygon analysis the Carstairs score was transferred from the OA to each postcode. Similar as in Italy and Slovakia, the Carstairs score in England was divided into 5 quintiles, 1 being the most affluent to 5 being the most deprived.

We used the distance from the point source (landfill site and/or incinerator) to estimate the exposed population. With very high level of confidence, we decided to use 3 km around incinerators (Elliott et al. 1996) and 2 km around landfill sites (Elliott et al. 2001) as the likely limit of the dispersion of emissions. For each plant (both incinerators and landfills), we defined increasing distance (1,2, and 3 Km) from the centre (the formal address of the plant) and evaluated the census blocks (or the postcode districts) that matched these surfaces. In several cases, the distribution of census blocks did not precisely fit the circle and the borders were cut so to more precisely count the population. A visual example of the method is provided in the graph below (Figure 2) for the incinerator of Modena in Italy and the next graph (Figure 3) illustrates the population distribution around the same plant. The validity of the method has been evaluated using individually geocoded data of the resident population in 4 areas of Emilia Romagna. It has been seen that the error range is between 1 and 10%.

Figure 2. Schematic representation of the definition of the population at risk on the basis of census blocks of residence. Incinerator of Modena, Italy.

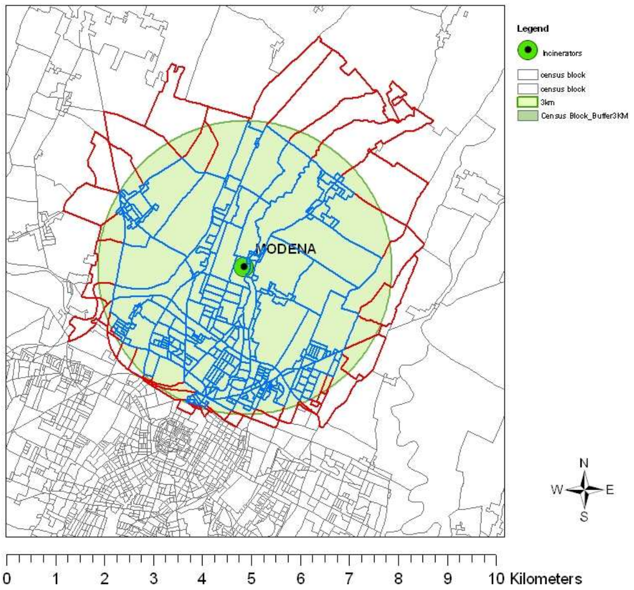
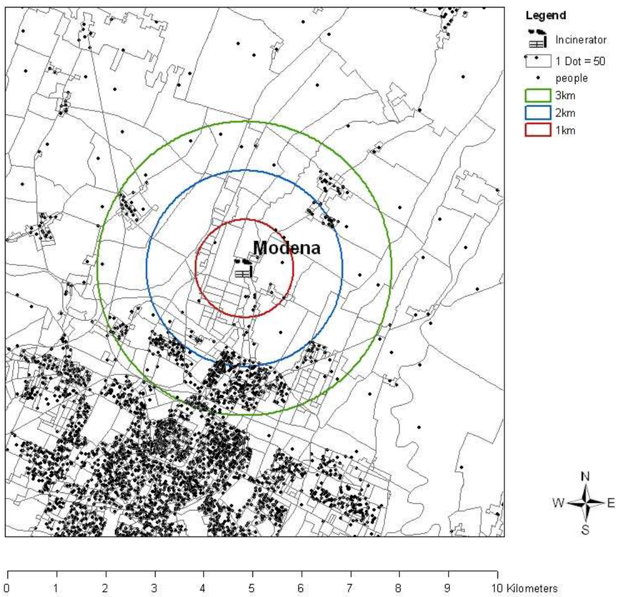


Figure 3. Population at risk on the basis of census blocks of residence near 3 km from the incinerator of Modena, Italy.

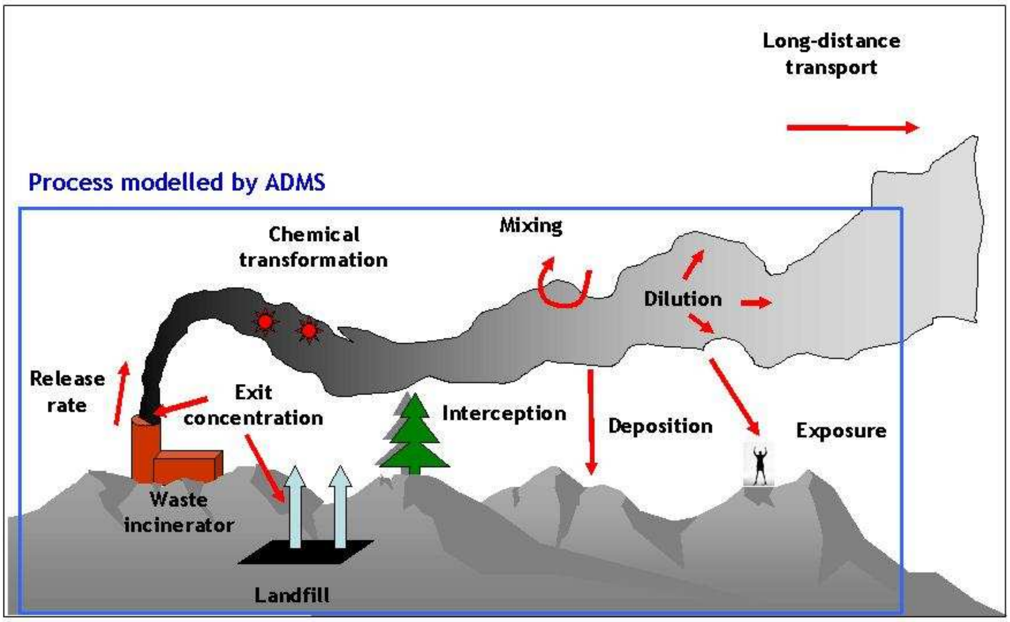


For each circle surrounding the plants, we estimated population size by gender, age, and socioeconomic status. We collected and stored the geographical variables using ArcView 9.2 Desktop GIS for Mapping, Data Integration, and Analysis (Environmental Systems Research Institute, Inc. (ESRI), Redlands, California, USA).

2.3.3 Methods for exposure assessment: local air dispersion modelling

In many point source epidemiological studies, distance has been used as a proxy measure of exposure as it provided quick and inexpensive estimates. On the other hand, this method has limitations that may result in exposure misclassification. Atmospheric dispersion modelling represents another approach in exposure assessment that may be more accurate if compared to the distance-based method. Point source characteristics, meteorological conditions and topographical features can be considered in dispersion modelling and used in health impact assessment. The figure below (Figure 4) is a schematic representation of the methodology.

Figure 4. Schematic representation of the methodology used in ADMS-Urban.



Local air dispersion modelling has been used for the calculation of increased pollutant concentrations (particulate matter, PM_{10}, and Nitrogen dioxide, NO_2) within 3 km from the waste incinerators in the three countries.

Dispersion modelling for incinerators was based on the national information on incineration census, actual waste throughput data and meteorological data. We have focused on the emissions from the waste gas stack.

We have used the Atmospheric Dispersion Modelling System (ADMS-Urban) developed by CERC and England Meteorological Office (CERC 1999) for modelling dispersion at the local scale for 40 incinerators in Italy, 2 in Slovakia and 11 in England. The model uses an up-to-date understanding of the atmospheric boundary. This is described by boundary layer depth and the Monin-Obukhov length, rather than by the Pasquill stability categories. Meteorological data requirements include temperature (°C), wind speed (m/s), wind direction (°), precipitation (mm), cloud cover (oktas), relative humidity (%), boundary layer height (m), and surface sensible heat flux (W/m2). We have used official meteorological data available from the nearest airport. Usually 2001 meteorological data were used.

ADMS-Urban is set up to model the pollutants NOx, NO_2, VOC, SO , CO, Benzene, Butadiene, PM10, and TSP, but may be used for other pollutants as well, including dioxins and PCBs. Only results for PM10 and NO2 have been used in the 1st pass assessment. Technical parameters necessary for modelling incinerators as a point source include stack height (m), stack diameter (m), exit velocity (m/s), emission rate (m3/s), exit temperature (°C) and location of the stack. For PM10 and NOx we have used emission rates based on national limits derived from current EU legislation, namely daily emission rates of 10 and 200 mg/Nm3, respectively. However, since actual emissions could be estimated from Italy and England, we conducted an additional analysis using real data.

Table 3 shows the difference between the actual emission rates and the emission rates based on the limit values plus the effect this has on modelled concentrations across the population (population-weighted) in England and Wales within 3 km of the incinerators operational in 2001. The actual emission rates are about half the limit rates for NOx and about 10 tims less in case of PM. This effect translates directly to the concentrations which see a similar reduction for both Nox and PM.

Table 3: Effect of actual and limit emission rates on population weighted concentration across 11 incinerators in England.

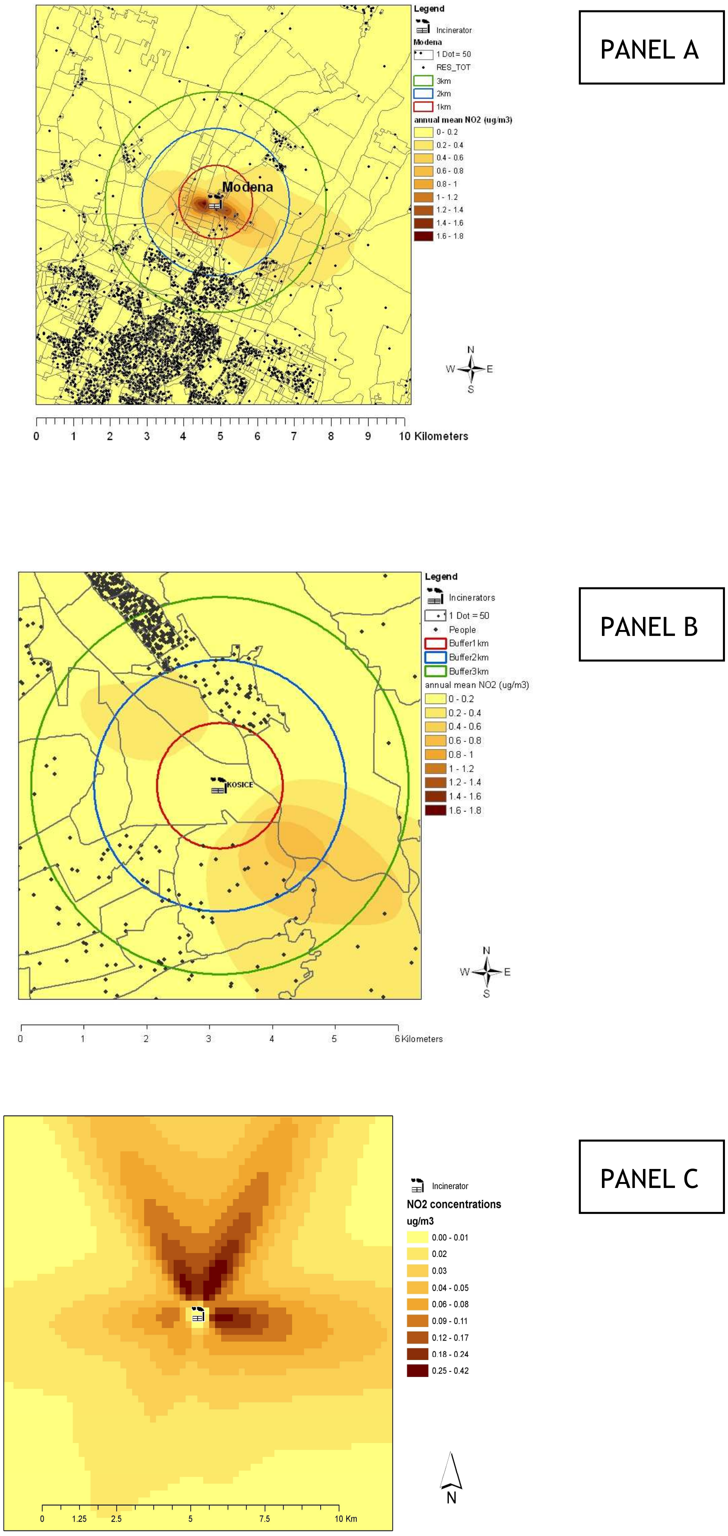
	Limit PM	Limit NOx	Actual PM	Actual NOx
Emission rates (g/s)				
Mean	0.78	15.71	0.081	6.46
Standard deviation	0*	0*	0.022	4.17
Conc (ug/m3)				
Mean	0.01	0.30	0.00	0.15
Standard deviation	0.01	0.21	0.00	0.13

* No standard deviation for limit values, as the same limit emission rates were used for all 11 incinerators

A number of incinerators in both Italy and England are located in hilly terrain. ADMS-Urban contains a hill module which takes into account the surrounding terrain when modelling the dispersion. Terrain data was therefore obtained for both these countries. For England the Ordnance Survey PANORAMA ^{TM} Digital Terrain Model was used to obtain surface heights for 50x50m cells up to 10km away from 8 of the 17 incinerators. For Italy the terrain data was collected from the Italian Environmental Protection Agency for 35 of the 40 incinerators.

ADMS air pollution dispersion model have provided "contours" of additional concentrations of PM and NO2 for the incinerators. These output files (one per country) have been transferred into the GIS system. The population database at the smallest available unit (i.e. census block or postcode district) for the given radius of 3 km has been added to the GIS as another data layer. Using an overlay function in GIS, the population data was combined with the air pollution concentration data with a grid of 200 meters. In this way, different statistics regarding population-weighted exposure levels have been estimated according to gender, age and socioeconomic status. Figure 5 illustrates, with examples from the three countries, the results of the dispersion model for NO2 around the Modena (Italy) incinerator (panel a), Kosice in Slovakia (panel b), and Stoke-on-Trent in England (panel c). We have very high confidence that this approach was able to accurately estimate population exposure.

Figure 5. Results of the air pollution dispersion models for NO2 for the area of Modena (panel A) (IT), Kosice (SK) (panel B) and Stoke-on-Trent (EN) (panel C).



2.3.4 Large Scale Air Dispersion Modelling

Large scale air dispersion modelling has been used to calculate increased pollutant concentrations in large areas. Modelling was performed using EcoSenseWeb, an integrated computer system developed for the assessment of environmental impacts and resulting external costs from electricity generation systems and other industrial activities. It is based on the Impact Pathway Approach (IAP) developed in the ExternE-Project (ExternE: www.ExternE.info). More details are presented in the full report in Appendix 5. Various modules of the EcoSenseWeb system refer to a so-called "local" and "regional" range analysis. The concept of local and regional range analysis results from the need of performing a European-wide (regional) analysis based on an operational amount of data, but also to take into account the spatial distribution of concentrations and receptors at a high resolution within the highly affected area close to the source of emissions. Models and data are provided in such a way that the standard impact assessment includes a local range analysis based on a 10 x 10 km2 EMEP grid (see Appendix 5), covering an area of 10 x 10 grid cells (i.e. 10,000 km2), with the source, e.g. an incinerator located in the centre of the local region.

Local range analysis: The Industrial Source Complex Model (ISC), a Gaussian plume model developed by the US-EPA, was used. The ISC is used for transport modelling of primary air pollutants (SO2, NOx, particulates) on a local scale (100 km x 100 km around the power plant site). EcoSense provides a short-term version of the model which uses hourly site-specific meteorological data. These data are generated within the EcoSenseWeb.

The regional range analysis: The regional range analysis is based on the large EMEP-gridcells (2500 km2) and covers all of Europe.

Regional impact assessment is made with regional SR-receptor matrices, i.e. parameterised results of model runs with the EMEP/MSC-West Eulerian dispersion model. These complex model runs are based on certain emission scenarios and meteorological conditions, and a reduction of a pollutant by 15% for each source of emission within a corresponding sub-region. Europe is divided into 66 regions, i.e. some larger countries are subdivided into regions. A matrix is created for a 15% reduction of an airborne pollutant (e.g. NOx) within a country / sub-region of Europe based on meteorological conditions (e.g. in the year 2000) and background emissions of the year 2010 or 2020. This matrix contains the results in terms of concentrations of a primary (NOx) or secondary (nitrates and ozone, increased sulphates, etc.) air pollutants on the 50 km x 50 km EMEP grid. The chemical reactions are highly complex and difficult to predict precisely. For example, a reduction of NOx emissions leaves more background NH3 for reaction with background SO2, etc.

Based on the predicted concentration values, the exposure of different receptors is calculated.

2.4 Exposure-response relationships

The next step in the health impact assessment was to select or develop a suitable set of exposure-response functions that link indicators of exposure, like residence in the area at different distance from the source or individual pollutant concentrations, with specific health endpoints. According to the guidelines provided from WP 1.3, the exposure-response function may be a slope of a regression line with the health response as the dependent variable and the stressor as the independent variable. Alternatively, an exposure-response function may be reported as a relative risk (RR) or excess risk (ER) (relative risk minus one) of a certain health response for a given change in exposure. We have derived relative risks related to residence near landfills and incinerators from a systematic review of the literature and we have adapted these values to fit our purposes. Exposure-response functions related to the long term effects on mortality from PM $_{2.5}$ and PM_{10} and NO_2 have been derived from the extensive existing reviews of epidemiological and toxicological data performed in the context of WP 3.1 (Transport).

2.4.1 Systematic review of epidemiological studies on health effects of exposure to emissions from waste management

We have conducted a systematic review of epidemiologic literature (1986-2006) on health effects associated with collecting, recycling, composting, incinerating, and landfilling of municipal solid waste with the specific aim to derive appropriate relative risk estimates associated to various waste management technologies. The full report is reported as Appendix 6.

Briefly, a total of 28 papers concerning health effects in communities living in proximity to waste sites have been reviewed. The following health outcomes were considered: cancers, birth outcomes (congenital malformations, low birth weight, multiple births, and abnormal sex ratio of newborns), respiratory, skin and gastrointestinal symptoms or diseases.

For each paper, we have reported in appropriate tables (available upon request) study design, population characteristics, exposure measures, and the main results (incl. control for major confounders) with respect to the quantification of the health effects studied. For each study, we have evaluated the potential sources of uncertainty in the results due to design issues. In particular, the possibility that selection bias, information bias, or confounding could artificially increase or decrease the relative risk estimate has been noted. The overall evaluation of the epidemiological evidence regarding the process/disease association was made on the basis of the IARC (1999) criteria, and two categories were chosen: “Inadequate” when the available studies were of insufficient quality, consistency, or statistical power to decide the presence or absence of a causal association; “Limited” when a positive association was observed between exposure and disease for which a causal interpretation was considered to be credible, but chance, bias, or confounding could not be ruled out with reasonable confidence. In no case the category “sufficient evidence” could be used. In order to derive appropriate relative risks, we considered the set of studies providing the best evidence and assigned an overall level of confidence in the specific effect estimate, based on a standardized scale (very high, high, moderate, low, very low)[1].

The overall conclusions of the review are reported below. Table 4 illustrates the overall evidence and Table 5 summarizes the relevant figures for health effects related to landfills and incinerators that are most reliable. For each relative risk the distance from the source has been reported as well as the overall level of confidence.

[1] These different characterizing statements were considered a scale of expressions of confidence, assessing the chance that the specific effect estimate is correct: very high confidence at least 9 out of 10; high confidence about 8 out of 10; moderate confidence about 5 out of 10; low confidence about 2 out of 10; and very low confidence less than 2 out of 10.

Table 4. Summary of the overall epidemiologic evidence on municipal solid waste disposal: landfills and incinerators.

Health effect	Level of evidence	
	LANDFILLS	**INCINERATORS**
All cancer	Inadequate	Limited
Stomach cancer	Inadequate	Limited
Colorectal cancer	Inadequate	Limited
Liver cancer	Inadequate	Limited
Larynx cancer	Inadequate	Inadequate
Lung cancer	Inadequate	Limited
Soft tissue sarcoma	Inadequate	Limited
Kidney cancer	Inadequate	Inadequate
Bladder cancer	Inadequate	Inadequate
Non Hodgkin's lymphoma	Inadequate	Limited
Childhood cancer	Inadequate	Inadequate
Total birth defects	Limited	Inadequate
Neural tube defects	Limited	Inadequate
Orofacial birth defects	Inadequate	Limited
Genitourinary birth defects	Limited[1]	Limited[2]
Abdominal wall defects	Inadequate	Inadequate
Gastrointestinal birth defects[3]	Inadequate	Inadequate
Low birth weight	Limited	Inadequate
Respiratory diseases or symptoms	Inadequate	Inadequate

[1] Hypospadias and epispadias

[2] Renal dysplasia

[3] The original estimates were given for "surgical corrections of gastroschisis and exomphalos"

"*Inadequate*": available studies are of insufficient quality, consistency, or statistical power to decide the presence or absence of a causal association
"*Limited*": a positive association has been observed between exposure and disease for which a causal interpretation is considered to be credible, but chance, bias, or confounding could not be ruled out with reasonable confidence.

Table 5. Relative risk estimates for community exposure to landfills and incinerators

Outcome	Distance from the source	Relative Risk (Confidence Interval)	Level of confidence[2]
LANDFILLS			
Congenital malformations (Elliott et al, 2001)			
All congenital malformations	Within 2 km	1.02 (99% CI=1.01-1.03)	Moderate
Neural tube defects	Within 2 km	1.06 (99% CI=1.01-1.12)	Moderate
Hypospadias and epispadias	Within 2 km	1.07 (99% CI=1.04-1.11)	Moderate
Abdominal wall defects	Within 2 km	1.05 (99% CI=0.94-1.16)	Moderate
Gastroschisis and exomphalos[1]	Within 2 km	1.18 (99% CI=1.03-1.34)	Moderate
Low birth weight (Elliott et al, 2001)	Within 2 km	1.06 (99% CI=1.052-1.062)	High
Very low birth weight	Within 2 km	1.04 (99% CI=1.03-1.06)	High
INCINERATORS			
Congenital malformations (Cordier et al, 2004)			
Facial cleft	Within 10 km	1.30 (95% CI=1.06-1.59)	Moderate
Renal dysplasia	Within 10 km	1.55 (95% CI=1.10-2.20)	Moderate
Cancer (Elliott et al, 1996)			
All cancer	Within 3 km	1.035 (95% CI=1.03-1.04)	Moderate
Stomach cancer	Within 3 km	1.07 (95% CI=1.02-1.13)	Moderate
Colorectal cancer	Within 3 km	1.11 (95% CI=1.07-1.15)	Moderate
Liver cancer	Within 3 km	1.29 (95% CI=1.10-1.51)	High
Lung cancer	Within 3 km	1.14 (95% CI=1.11-1.17)	Moderate
Soft-tissue sarcoma	Within 3 km	1.16 (95% CI=0.96-1.41)	High
Non-Hodgkin's lymphoma	Within 3 km	1.11 (95% CI=1.04-1.19)	High

[1] The original estimates were given for "surgical corrections of.."
[2] The following scale for the level of confidence has been adopted: very high, high, moderate, low, very low.

Incinerators

Quantitative estimates of excess risk of specific cancers in populations living near solid waste incinerator plants were provided by Elliott et al. (1996). We have reported in Table 5 the estimates for all cancers, stomach, colon, liver, and lung cancer. The excess risk estimate for all cancers combined was 3.5% (95%CI=3-4%). However, the authors pointed out that there was an indication of residual confounding from socioeconomic status near the incinerators and a concern of misdiagnosis among registrations and death certificates for liver cancer. These aspects lowered the overall confidence of the results. On these bases, we scored the level of confidence of the risk estimates for these tumours as "moderate" with the exception of liver cancer (high confidence) since a reassessment of the possible misdiagnoses was made in the original study and the extent of residual confounding was lower for this form of cancer than for other neoplasms. In the Elliott et al (1996) study no significant decline in risk with distance for non-Hodgkin's lymphoma and soft tissue sarcoma was found. However, the studies of Viel et al (2000) and Floret (2003) conducted in France and the study from Comba et al (2003) in Italy provide some indications that an excess of these form of cancers may be related to emissions of dioxin from incinerators. In fact, a recent study by Zambon et al. (2007) clearly showed a significant increase in the risk of soft-tissue sarcoma, correlated both with the level and the length of environmental modelled exposure to dioxin-like substances emitted by waste incinerators. As a result, we provided effect estimates in Table 5 also for non-Hodgkin's lymphoma and soft tissue sarcoma as derived from the conservative "first stage" analysis conducted by Elliott et al (1996). We scored the level of confidence of this relative risk estimate as "high".

With regards to congenital malformations near incinerators, Cordier et al (2004) provided effect estimates for facial cleft and renal dysplasia, as they were more frequent in the "exposed" communities living within 10 km of the sites. Other reproductive effects, such as an effect on twinning or sex determination, have been described; however the results were inadequate.

Landfills

The epidemiological studies on cancer are not sufficient to draw firm conclusions regarding the health effects of living near landfills. The two studies from Goldberg et al (1999) and Pukkala et al (2001) are not consistent with regards to the cancer types, with the only exception of pancreatic cancer. The largest study conducted in England by Jarup et al (2002) does not suggest an increase for the cancer types that were investigated. For other chronic diseases, especially respiratory diseases, investigations are lacking with only one suggestive indication of an increased risk of asthma in adults (Pukkala et al, 2001) but with no replication of the findings. Overall, the evidence that living near landfills may be associated with health effects in adults is inadequate.

A different picture appears for congenital malformations and low birth weight where suggestive evidence exists of an increased risk for babies born to mothers living near landfill sites. The relevant results come from Elliott et al. (2001). Statistically significant increased risks were found for all congenital malformations, neural tube defects, abdominal wall defects, surgical correction of gastroschisis and exomphalos, and low and very low birth weight for births occurring in people living within 2 km from the sites. Several alternative explanations for the findings, including ascertainment bias and residual confounding, cannot be excluded. We concluded that Elliott et al (2001) provides quantitative effect estimates whose level of confidence can be considered as moderate. In addition, the effect on low birth weight has been confirmed in a complete study from Alaska (Gilbreath et al, 2006).

Conclusions

We found limited evidence of an association between living close to incinerators and landfills and some health effects. For incinerators, we decided to use the relative risk for all cancer (Elliott et al. 1996) as basis for impact assessment. For landfills, we decided to use the relative risk for congenital malformations and low birth weight (Elliott et al. 2001). We have a moderate level of confidence in these effect estimates.

2.4.2 Concentration-response relationships for PM_{10} and NO_2

The exposure – response function for PM10 and NO2 and "chronic mortality" have been summarized based on the available literature using criteria identified in WP 1.3 exposure-health effects of INTARESE. The work was performed in the WP 3.1 work package. The work was based on the use of already published meta-analyses and systematic reviews. We assumed a linear relationship between the air pollutants and associated health effects as most epidemiological studies on large populations have been unable to identify a threshold concentration below which ambient air pollutants has no effect on morbidity and mortality.

The following values were used:

RR= 1.06 (95%CI=1.03-1.09) increase in mortality for 10 ug/m^3 PM_{10}

RR= 1.06 (95%CI=1.04-1.08) increase in mortality for 10 ug/m^3 NO_2

2.5 Quantification of Health Effects

In the 1st phase, health effects were quantified for the baseline national policy scenarios for the year 2001. The current baseline scenario involves assumptions about the time period of operation of the incineration plants and a specific choice of the time of the evaluation. We assumed therefore that the incinerators operating in 2001 will be operating until 2020 and the health effects are estimated up to 2050. We believe that the choice of 2020 is realistic since these plants are in operation for a long time. The choice of 2050 guarantees enough time to fully evaluate chronic effects. For incinerators, cancer incidence “attributable” to exposure before 2001 (“past exposure”) was estimated (burden of disease non-modifiable by policy) as it is likely that it will continue to appear until 2050. In addition, cancer incidence “attributable” to exposure during 2001-2020 was estimated (“current exposure”) as these effects could be, at least in part, prevented. In addition, Years of Life Lost (YoLL) were estimated as attributable to current exposure (2001-2020) to PM and NO2 in the cohort of 2001 residents followed up to 2050. For landfills operating in 2001, we assumed that the emissions will last up to 2030 (an assumption in agreement with the available knowledge that landfilled biodegradable waste starts to emit gas a few years after deposit and continues to do so for several decades) and the health effects, in terms of congenital anomalies and low birth weight, are constant throughout this period. Of course, this statement assumes no improvement in the technology of gas recovery.

2.5.1 Background health statistics for quantification

Background sex-age specific cancer incidence data for the three countries were retrieved. In particular, data from the Italian cancer registries (www.registri-tumori.it) were downloaded for Italy, from the National Cancer Registry of the Slovak Republic (http://www.nor-sk.org/) for Slovakia and from Cancer Registration Statistics England 2001 (www.statistics.gov.uk) for England.

National mortality statistics were available from the national Institute of statistics (http://demo.istat.it/) for Italy, from The Statistical Office of the Slovak Republic (www.statistics.sk)for Slovakia and from the Office for National Statistics (ONS) for England.

Prevalence of congenital malformations at birth was derived from the Annual Report (data for 2000) of the International Clearinghouse for birth defects monitoring system (www.icbd.org) for Italy (all registers) and England, and from The Statistical Office of the Slovak Republic (www.statistics.sk) for Slovakia.

2.5.2 Estimating cancer incidence near incinerators

The basic formula to compute the number of cancer cases attributable to an incinerator is:

$AC = Rate_{unex} * ER * Pop_{exp}$

where AC = the attributable cancer incidence

$Rate_{unex}$ = background incidence rate in the general population

ER = excess risk in the exposed population (relative risk – 1)

Pop_{exp} = number of exposed people

As already indicated, we decided to use the excess risk for all cancer from Elliott et al. (1996). Although the possibility to infer causality from this study is limited (due to the limitations discussed above), the estimate is a unique starting point for our assessment.

We also assumed that the excess risk is not constant over time, but varies for a specific individual of the population at a given age and specific time as a function of various characteristics: level of attained cumulative exposure, latency since first exposure and latency since cessation of exposure (if any). We therefore assumed a theoretical model of cancer occurrence and imputed the varying excess risk around different incinerators, as a function of the different characteristics of the plant and of the nearby population. The methods are fully described in Appendix 7. Briefly, we modified the excess risk for overall cancer incidence estimated by Elliott et al (1996) (i.e. 3.5% for people exposed at incinerators operating before 1980, assuming 20 years of exposure) as a function of cumulative exposure (with exposure coefficients varying with time), latency since first exposure and latency since cessation of exposure.

Thus, for a given age group (*a_i*):

$$ER_{a\_i} = RER * (CE_{a\_i} / 20) * Ls * Lc \qquad (6),$$

Where

ER_{a_i} = the estimated excess risk of cancer incidence

RER =the reference excess risk as estimated from Elliott et al (1996) (3.5% increase for exposure of 20 years to incinerators operating before 1980).

CE_{a_i} = cumulative exposure

Ls =latency since start of exposure

Lc =latency since cessation of exposure r

And for a given age group (*a_i):*

$$AC_{a\_i} = ER_{a\_i} * Rate_{un\exp} * Pop_{\exp} \qquad (7),$$

where

ACa_i =attributable cancer incidence

ER a_i =excess risk of cancer incidence

$Rate_{unex}$ =background incidence rate in the general population

Pop_{exp} =number of exposed people

This algorithm was applied to the estimated 2001 population (by sex and age) living within 3 km from each specific incinerator to estimate the number of excess cancer cases in 2001-2050 attributable to exposures before 2001 and during 2001-2020.

In Appendix 7, we illustrate the basic assumptions and we show how the excess risk during the evaluation period 2001-2050 varies in relation to time since the start of the operation of the plant and the time since cessation. The key assumption we made (motivated by measured data) is that the exposure levels during 1980-1989, 1990-2000 and after 2000 were 0.8, 0.2, and 0.05 of those occurring before 1980. Although we are highly confident about the scores we gave to the exposure levels (as they are confirmed from measured data and are reflected in the legislation), we have moderate confidence in the overall procedure of estimating cancer cases.

2.5.3 Estimating years of life lost (YoLL)

On the assumption to follow up until 2050 the entire 2001 population living close to incinerators in the three countries, and that their mortality rate was similar to that of the national population in 2001, we estimated Years of Life Lost attributable to PM10 and NO_2 exposure as derived from the air dispersion model. In particular, we assumed that the impact of PM10 and NO2 will be felt only during 2001-2020. We have used the system of spreadsheets provided by the IOM institute (http://www.iom-world.org/pubs/IOM_TM0601.pdf?PHPSESSID=551b9ccae82ad1127a41db2c144d6d9a).

2.5.4 Estimating congenital anomalies and low birth weight near landfills

With a moderate level of confidence, we have assumed that the only health impacts on populations living near landfills are congenital malformations and low birth weight. We acknowledge our ignorance about additional health effects. The simple algorithm below was used to calculate the number of congenital malformations and babies of low birth weight attributable to residence near landfills. As already indicated, the time of the evaluation is 2001-2030 on the assumption that gas emissions from landfills last 30 years.

$$AC = Rate_{unex} * ER * Pop_{exp}$$

where AC = the attributable cases of malformation

$Rate_{unex}$ = background prevalence rate in the general population

ER = excess risk in the exposed population (relative risk – 1)

Pop_{exp} = number of exposed newborns

The excess risk was derived from the literature referenced above and the number of newborns was estimated with GIS. This algorithm was applied to the estimated 2001 population of newborns living within 2 km from each specific landfill.

2.5.5 Estimating the health impact of incineration on a European scale

According to the Impact Pathway Approach (IPA), the physical impact of the receptors can be calculated by multiplying the concentration in each grid cell with the number of people and with a factor according to the concentration-response relationship per unit of concentration increment. The impact over all of Europe can then be understood. The concentration increment is taken from the large scale dispersion modelling.

The Concentration Response Functions (CRF) for human health impacts due to classical air pollutants are reported in Appendix 5. These are the most important and reliable concentration response functions (core) derived in the NEEDS project (Torfs et al, 2007). The corresponding monetary values demonstrate the different severity of the endpoints. Since the probability of the less severe endpoints is much higher they also contribute considerably to the total damage. Nonetheless, the reduced life expectancy (YOLL, Years of life lost) is the most important endpoint. For the purpose of this case study on waste incineration only the YOLL of classical air pollutants are assessed and reported in this section on large scale modelling.

Three CRFs are considered to evaluate the number of YOLL.

Life expectancy reduction due to PM2.5

The increased mortality of infants due to PM10 and

The increased mortality due to ozone

The CRFs for PM2.5 calculates the YOLL directly. With regard to increased mortality of infants, 80 YOLL per case is assumed, with regard to increased mortality due to ozone 0.75 years per case are assumed.

The population of Europe (*SEDAC, 2006)* is allocated to a 50 x 50 km2 grid. For each grid the physical impacts as years of lifetime lost (YOLL) due to one year of operation and corresponding annual emissions is calculated. Most of the YOLL will occur not in the same year but in the future, approximately up to 30 years later (2030).

2.6 Quantification of external costs

A complete evaluation of the external costs of waste management will be performed in the second phase, on the basis of a comprehensive strategy of the INTARESE project.

2.7 Sensitivity analyses

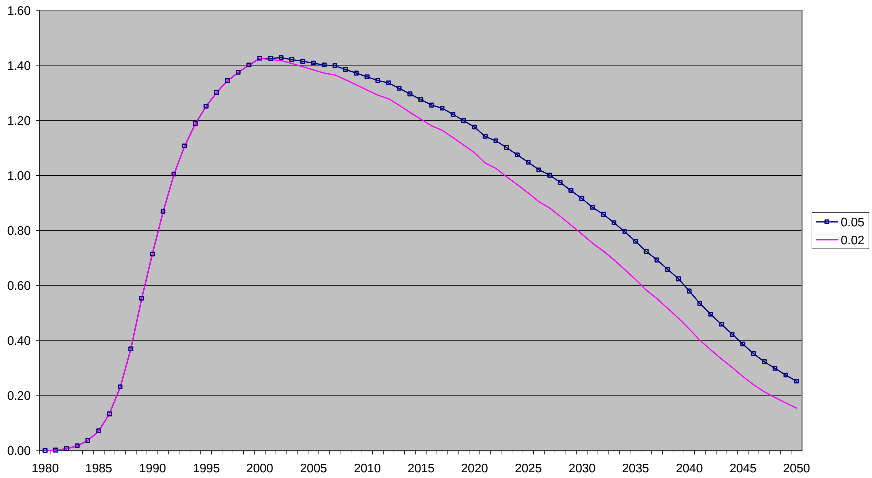
We have planned several sensitivity analyses to evaluate the robustness of the results.

2.7.1 Sensitivity analysis regarding cancer incidence attributable to incinerators operating in 2001-2020

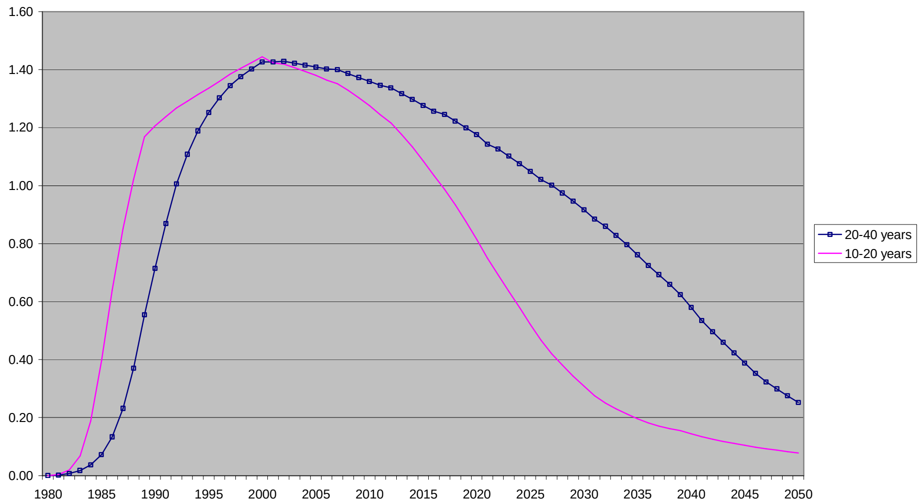
In our model to estimate cancer incidence attributable to incinerators operating in 2001-2020, three assumptions were critical: the coefficient we assigned to the exposure level in 2001-2020, i.e. 0.05; the form of the relationship with latency since exposure began and the form of the relationship with time since exposure ceased. While the first assumption is related to emission control measures, the other two are connected with the biological relationship between exposure to carcinogens and occurrence of cancer. We modified these assumptions in order to perform a sensitivity analysis as follows.

1. We modified our original assumption that incinerators after 2001 emit twentyfold less air pollutants than old incinerators operating before 1980 with a more radical coefficient of 0.02 (fifty-fold reduction). The first graph below shows, for an incinerator operating since 1980, a slight change in the Excess Risk (ER, Y axis) after 2001.
2. We modified our original assumption that the full effect on cancer expression is reached 20 years since first exposure and has a plateau lasting up to 40 years. We assumed a faster slope of the latency function and a shorter duration of the plateau: the peak of the effect is already reached at 10 years and declines after 20 years. The second graph below shows, for an incinerator operating since 1980, a shift to the left in the Excess Risk (ER, Y axis) before 2001 and a rapid decline since that date.
3. We modified our original assumption that the effect declines smoothly after cessation of exposure, reaching a minimum after 20 years applying a stronger effect of time since cessation reaches a null effect 10 years after closure. The third graph below shows, for an incinerator operating since 1980, a shift to the left in the Excess Risk (ER, Y axis) after cessation of exposure.

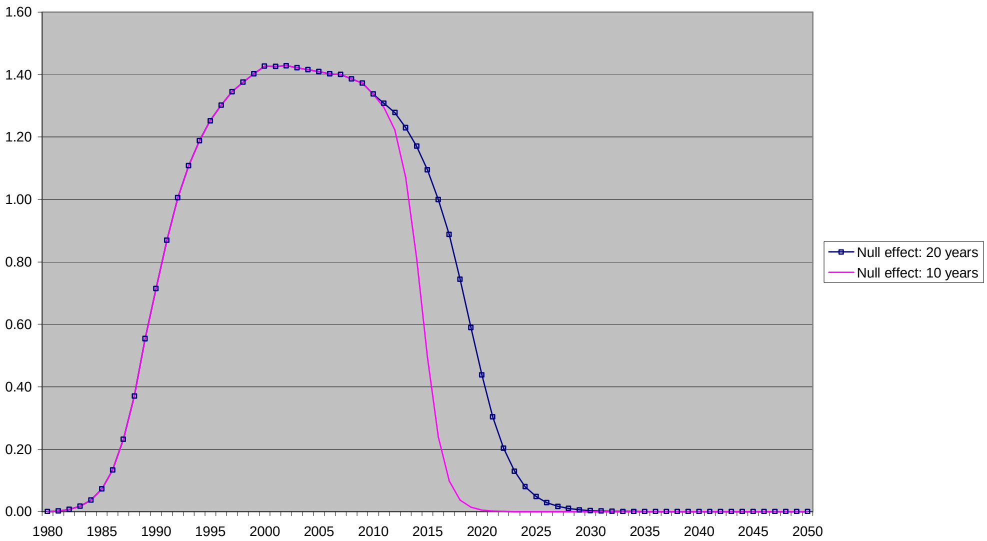
Sensitivity analysis 1: change in the coefficient for exposure in 2001-2020



Sensitivity analysis 2: change in the coefficient for latency since first exposure



Sensitivity analysis 3: cessation in 2010, latency since cessation



2.7.2 Sensitivity analysis regarding calculations of YoLL

We planned to evaluate the sensitivity of using a different approach for calculating YoLL with the lifetable approach.

1. To change our assumption that the impact on the force of mortality applies only during current exposure (2001-2020) and then abruptly ends. We wish to allow the mortality effect of PM10 and NO2 to slightly increase during 2001-2005 and then smoothly decline up to 10 years after cessation of exposure.
2. We wish to divide the impact of incinerators on life expectancy into two components: effect due to cancer and effect due to other diseases. For this purpose, we substitute in the life table analysis the overall coefficient with one specific for cancer based on our estimated cancer excess risk for "current exposure" and the other based on the modeled NO2/PM10 concentrations.

2.7.3 Sensitivity analysis regarding effect of landfills on congenital malformations.

1. We modified our original assumption that the effect on congenital malformation is similar to what has been found in the original study in England (Elliott et al. 2001). We assumed that the emissions from landfills have been more controlled during years that had with a weaker effect (0.5) on newborns.

3 Results

3.1 Waste Generation and Management in Italy, Slovakia and England

A short description of waste management in the three countries in 2001 is reported in Appendix 3. Briefly, Table 6 and Figure 6 illustrate the basic statistics of waste management in Italy, Slovakia and England in 2001.

The amount of MSW produced in Italy was 31.94 million of tonnes (Mtonnes), which corresponds to 560 kilograms per inhabitant. Thirty percent consisted of compostable materials (9.59 Mtonnes), 54% of recyclable materials (17.25 Mtonnes) and the remaining 16% consisted of unsortable materials (5.11 Mtonnes). Regarding MSW treatment plants, there were 212 composting plants with a total annual potential treatment capacity of 4.26 Mtonnes; 65 Mechanical-biological treatment (MBT) plants with a total annual potential treatment capacity of 6.78 Mtonnes; 40 incineration plants with a total annual potential treatment capacity of 3.8 Mtonnes; 619 landfills with a total storage volume of about 200 million m^3 (150 Mtonnes) and a residual storage volume of 100 million m^3 (75 Mtonnes); 2,400 recycling plants (estimate) with a total annual potential treatment capacity of 15 Mtonnes. In 2001 about 56% of Italian MSW was directed to landfills, which were the most used facilities, and recycling and composting accounted for 16% and 8% of MSW.

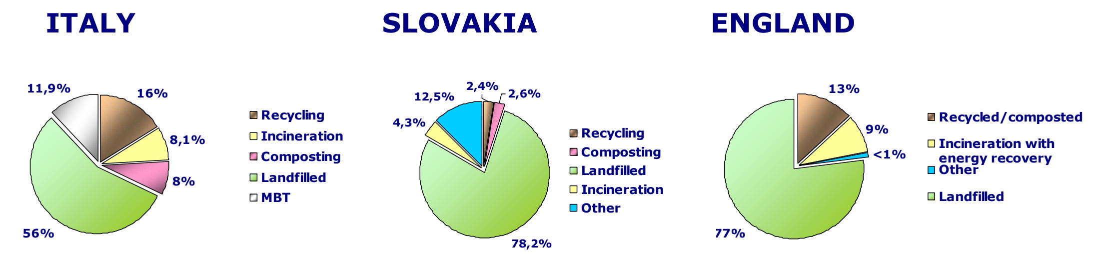
Concerning Slovakia, after consultation with the Statistical Office of the Slovak Republic, data on MSW production and management in 2001 were substituted by data from 2002. The main reason for this substitution was a change in the reporting system that was realized in Slovakia during 2001, including implementation of the new law "Act on Waste no. 223/2001" and re-allocation of the responsibilities for reporting waste production and management. One of the additional important differences was reporting of the waste coming from septic tanks – based on liquid vs. dry matter. The amount of MSW produced in Slovakia in 2002 was 1.52 million of tonnes (Mtonnes), which corresponds to 283 kilograms per inhabitant. From this total, 12% (0.18 Mtonnes) was recovered/treated (2.4% recycled, 2.6% composted, 6% recovered as energy and 1% treated by other methods) and 88% was disposed (4.3% incinerated, 78.2% landfilled and 5.5 disposed by other methods). The number of landfills decreased from 165 to 154 between 2001 and 2002, while the number of incinerators remained the same - 2 incinerators for MSW.

The amount of MSW produced in England was 28.8 million of tonnes (Mtonnes), which corresponds to 587 kilograms per inhabitant. The majority of the MSW was landfilled (22 Mtonnes, or 77%), followed by recycling and composting (3.7 Mtonnes, 13%) and 9% of MSW was incinerated (2.6 Mtonnes).

Table 6. Waste Generation and Management in Italy, Slovakia and England

DESTINY	ITALY		SLOVAKIA (2002)		ENGLAND	
	Thousands tons	[%]	Thousands tons	[%]	Thousands tons	[%]
Landfill	17,910	56	1,192	78	22,180	77
Incineration	2,590	8	65	4	2,590	9
Recycled/composted	7,650	24	76	5	3,740	13
Other	3,790	12	191	13	290	1
TOTAL	**31,940**	**100**	**1,524**	**100**	**28,800**	**100**
MSW generation per capita/kg	560		283		587	

Figure 6. Waste management in Italy, Slovakia and United Kingdom



Source: APAT - Agenzia per la Protezione

Figure 7-9 illustrates the map of the incineration plants and landfills in Italy, Slovakia, and England, respectively.

Figure 7. Map of incinerators and landfills in Italy

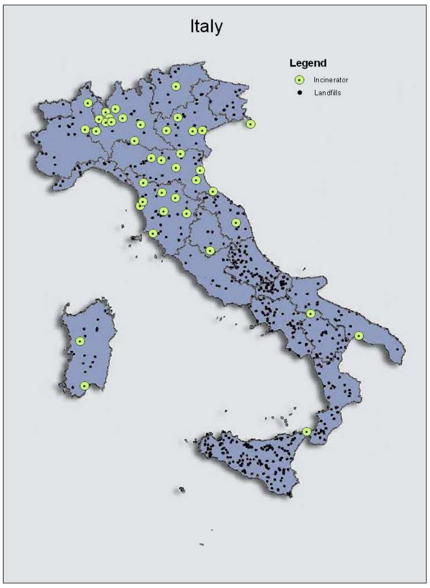


Figure 8. Map of incinerators and landfills in Slovakia

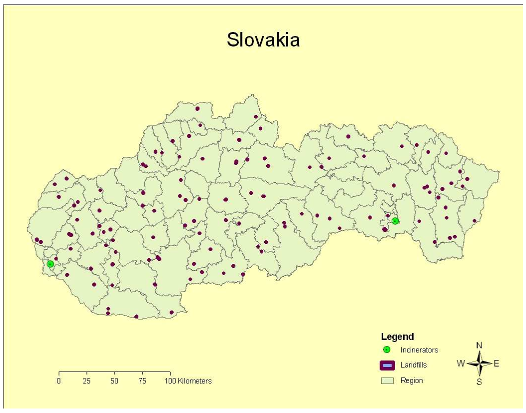
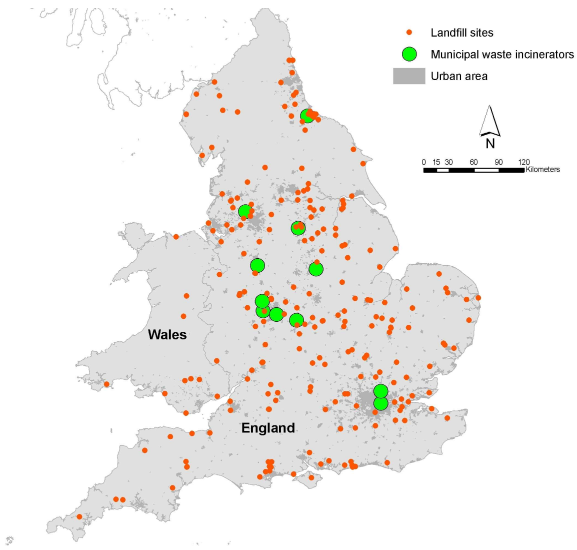


Figure 9. Map of incinerators and landfills in England



Tables 7-9 list the characteristics of the incinerators in the three countries, including year of construction and fumes capacity and stack height. Table 10-12 illustrates the characteristics of the landfills.

Table 7. Characteristics of incinerators in Italy, 2001

Area	Region	Place	Year of start operation	Fumes capacity (Nmc/h) line 1	line 2	line 3	Exit Temp. (°C)	Chimney height (m)
NORTH	PIEMONTE	VERCELLI	1991	80000	80000	80000	85	35
NORTH	LOMBARDIA	COMO	1967	90000			86	60
NORTH	LOMBARDIA	DESIO	1976	38000	38000		147	47
NORTH	LOMBARDIA	CREMONA	1997	37000	48000		125	60
NORTH	LOMBARDIA	BRESCIA	1998	187000	187000	220000	133	120
NORTH	LOMBARDIA	BUSTO ARSIZIO	2000	67800	67800		133	120
NORTH	LOMBARDIA	MILANO	2000	108000	108000	108000	136	120
NORTH	LOMBARDIA	PARONA	2000	**45000**	**45000**		**130**	**80**
NORTH	LOMBARDIA	DALMINE	2001	55000	55000		167	80
NORTH	LOMBARDIA	SESTO S.GIOVANNI	2001	22000	22000	22000	115	70
NORTH	TRENTINO ALTO ADIGE	BOLZANO	1988	126000	126000		150	50
NORTH	VENETO	PADOVA	1965	**45000**	**45000**		**130**	**80**
NORTH	VENETO	SCHIO	1982	12000	19000	36000	135/100/150	40
NORTH	VENETO	FUSINA	1998	**45000**	**45000**		**130**	**80**
NORTH	VENETO	CA' DEL BUE	1999	68000	68000		130	60
NORTH	FRIULI VENEZIA GIULIA	TRIESTE	2000	52600	52600	49000	120	100
NORTH	EMILIA ROMAGNA	REGGIO EMILIA	1968	50400	50400		170	40
NORTH	EMILIA ROMAGNA	BOLOGNA-granarolo	1973	45000	45000	45000	131	80
NORTH	EMILIA ROMAGNA	CORIANO	1976	35000	35000	35000	145	40
NORTH	EMILIA ROMAGNA	FORLI	1976	31000	31000		110	60
NORTH	EMILIA ROMAGNA	MODENA	1980	32500	32500	50000	>100	80
NORTH	EMILIA ROMAGNA	FERRARA CANALB.	1993	45000			61	82
NORTH	EMILIA ROMAGNA	RAVENNA	2000	55000			102	60
CENTER	TOSCANA	LIVORNO	1974	**45000**	**45000**		**130**	**80**
CENTER	TOSCANA	MONTALE AGLIANA	1976	50000			140	40
CENTER	TOSCANA	CASTELNUOVO di Garfagnana		1977	15000		150	30
CENTER	TOSCANA	POGGIBONSI	1977	28425			146	40
CENTER	TOSCANA	OSPEDALETTO	1980	**45000**	**45000**		**130**	**80**
CENTER	TOSCANA	VALMADRERA	1981	70 000	70 000		82	52
CENTER	TOSCANA	RUFINA	1995	**45000**	**45000**		**130**	**80**
CENTER	TOSCANA	AREZZO	2000	**45000**	**45000**		**130**	**80**
CENTER	TOSCANA	SCARLINO	2000	**45000**	**45000**		**130**	**80**
CENTER	UMBRIA	TERNI	1998	20000	20000		60	40
CENTER	MARCHE	TOLENTINO	1989	**45000**	**45000**		**130**	**80**
SOUTH	PUGLIA	STATTE	1976	25000	25000		150	40
SOUTH	BASILICATA	MELFI	1999	45000	55000		140	50
SOUTH	CALABRIA	MERGOZZO	1960	18000	18000		105	50
SOUTH	SICILIA	MESSINA	1979	**45000**	**45000**		**130**	**80**
SOUTH	SARDEGNA	MACOMER	1994	21000	21000		160	40/45
SOUTH	SARDEGNA	MACCHIAREDDU	1995	**45000**	**45000**		**130**	**80**

Values in bold have been inputed

Table 8. Characteristics of incinerators in Slovakia, 2001

Area	Place	Year of start operation	Fumes capacity (Nmc/h) line 1	Fumes capacity (Nmc/h) line 2	Exit Temperature (°C)	Chimney height (m)
Bratislava	BRATISLAVA	1977	90000	178000	155/205	120
Kosice	KOSICE	1989	72000		150/240	105

Table 9. Characteristics of incinerators in England, 2001

Area	Region	Place	Exit Temperature (°C)	Chimney height (m)
England	North West	Bolton		60
England	London	Edmonton		75
England	East Midlands	Eastcroft		91
England	West Midlands	Coventry		75
England	Yorkshire and Humber	Sheffield		76
England	London	Lewisham	140	100
England	West Midlands	Tyseley		75
England	West Midlands	Stoke-on-Trent		75
England	North East	Stockton-on-Tees		75
England	West Midlands	Dudley		75
England	West Midlands	Wolverhampton		76

Table 10. Characteristics of landfills in Italy, 2001

Area	Region	Number of legal landfills	Urban wastes landiflled (tons) in 2001*	Number of geocoded landfills
NORTH	Piemonte	22	1647132	16
NORTH	Valle d'Aosta	1	57706	0
NORTH	Lombardia	10	1503737	0
NORTH	Trentino Alto Adige	15	272282	0
NORTH	Veneto	21	1166733	0
NORTH	Friuli Venezia Giulia	12	236753	0
NORTH	Liguria	16	871359	0
NORTH	Emilia Romagna	29	1690238	26
CENTER	Toscana	31	1087963	31
CENTER	Umbria	7	391957	0
CENTER	Marche	19	571162	0
CENTER	Lazio	11	2620620	0
SOUTH	Abruzzo	58	504312	27
SOUTH	Molise	40	131451	0
SOUTH	Campania	56	1655569	18
SOUTH	Puglia	22	1724564	0
SOUTH	Basilicata	28	179447	0
SOUTH	Calabria	48	731497	0
SOUTH	Sicilia	156	2244087	0
SOUTH	Sardegna	17	714291	0
Total		619	20002860	118

Table 11. Characteristics of landfills in Slovakia, 2001.

Area (No of landfills)	Region	Number of legal landfills	Urban wastes landiflled (tons) in 2001*	Number of geocoded landfills
Bratislava (7)	Malacky	2	185764	2
(Bratislavský kraj)	Pezinok	3	27247	2
	Senec	1	33627	1
	Bratislava IV	1	3177	1
Trnava (26)	Trnava	4	96145	2
(Trnavský kraj)	Hlohovec	6	12073	5
	Piešťany	1	19820	1
	Senica	4	31563	3
	Skalica	3	17692	2
	Galanta	4	33387	2
	Dunajská Streda	4	35585	3
Nitra (23)	Komárno	3	26207	3
(Nitriansky kraj)	Nitra	3	5887	2
	Levice	6	62794	5
	Šaľa	3	13971	3
	Nové Zámky	5	68723	2
	Topoľčany	1	0	0
	Zlaté Moravce	2	9575	1
Trenčín (17)	Myjava	3	24728	1
(Trenčiansky kraj)	Trenčín	1	0	0
	Ilava	2	93664	2
	Nové Mesto n. V.	1	0	0
	Partizánske	2	20005	3
	Bánovce n. Bebr.	1	29105	2
	Púchov	2	17560	1
	Považská Bystrica	1	18844	1
	Prievidza	4	37388	3
Žilina (22)	Žilina	3	115000	2
(Žilinský kraj)	Liptovský Mikuláš	5	70931	3
	Martin	3	37030	1
	Námestovo	1	7850	1
	Bytča	1	8000	1
	Dolný Kubín	2	50914	2
	Kysucké N. Mesto	1	790	1
	Ružomberok	1	0	2
	Turčianske Teplice	1	0	0
	Tvrdošín	3	4402	1
	Čadca	1	22605	1
Banská Bstrica (29)	Banská Bystrica	2	36721	1
(Banskobystrický kraj)	Banská Štiavnica	1	8177	1
	Brezno	2	29734	2
	Detva	1	4584	1
	Krupina	2	6732	2
	Lučenec	2	25813	1
	Poltár	2	6941	2
	Revúca	3	0	1
	Rimavská Sobota	2	2746	2
	Veľký Krtíš	6	8256	3
	Zvolen	1	0	1
	Žiar n. Hronom	5	91471	4
Košice (16)	Košice I	1	71727	1
(Košický kraj)	Košice II	1	5875	1
	Košice - okolie	1	3170	1
	Michalovce	5	323221	5
	Trebišov	2	16816	2
	Rožňava	3	14392	3
	Spišská Nová Ves	2	0	0
	Sobrance	1	1503	1
Prešov (21)	Prešov	1	0	1
(Prešovský kraj)	Bardejov	2	9038	1
	Humenné	4	36729	3
	Kežmarok	1	44497	2
	Medzilaborce	1	2078	1
	Poprad	1	0	1

	Sabinov	2	1504	1
	Snina	2	7277	2
	Stará Ľubovňa	1	12061	1
	Stropkov	1	3994	1
	Svidník	1	2774	1
	Vranov n. Topľou	4	10365	3
TOTAL		161	2030249	121

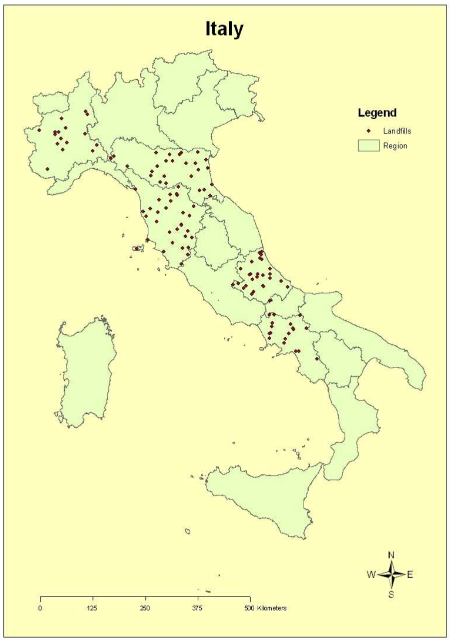
*only for geocoded landfills

Table 12. Characteristics of landfills in England 2006 (representative for 2001 situation).

Region	Number of legal landfills	Urban wastes landiflled (tons) in 2006*	Number of geocoded landfills
North East	37	4,846	19
North West	73	9,509	31
Yorks & Humber	78	6,791	26
East Midlands	70	6,976	28
West Midlands	40	5,775	23
East of England	75	11,197	39
London	7	1,977	3
South East	85	11,979	31
South West	70	5,887	32
England	**535**	64,937	**232**

Figure 10 illustrates the locations of the 118 Italian landfills for which complete geocoding was available

Figure 10. Landfills with geocoded location in Italy, 2001.



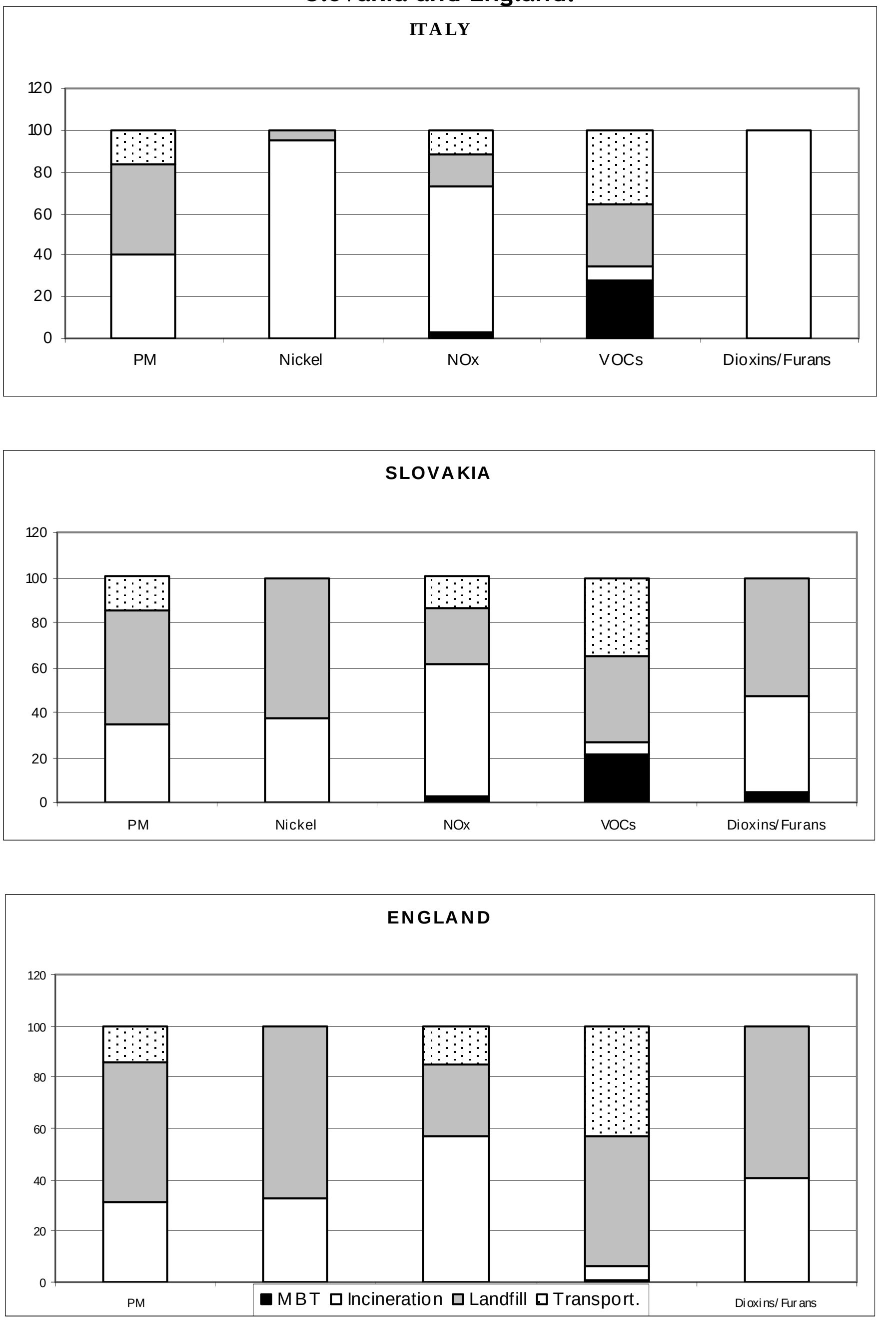
3.2 Quantification of Emissions of Pollutants from Waste Management in Italy, Slovakia and England

Table 13 presents the estimated amount of pollutants (total and per inhabitant) generated by waste treatment facilities. NOx, SO2, HCL, VOCs and PM are by far the largest emissions in the three countries. When emissions per inhabitant were considered, Italy had higher values than the other countries for arsenic, mercury and dioxins whereas generally lower values were found for all pollutants in Slovakia. Figure 11 shows, for each country, the proportion of the total emissions for each specific process. For PM, the largest contribution comes from incineration, landfills and (to a lesser extent) transport. For nickel and dioxins the largest contribution is from incinerators (especially in Italy) and landfills. NOx pollution derives from incineration, landfills and transportation, VOCs derive from landfilling, transport and MBT.

Table 13. Total emissions due to management of MSW in three EU countries (Total and per inhabitant).

	Italy	Slovakia	England	Italy	Slovakia	England
Compound	Total (Tonnes)	Total (Tonnes)	Total (Tonnes)	Per inhab. (grams)	Per inhab. (grams)	Per inhab. (grams)
PM	274.1	14.98	282.55	4.81	2.77	5.75
Cadmium	0.3	0.02	0.30	0.005	0.003	0.006
Nickel	3.4	0.02	0.35	0.060	0.003	0.007
Arsenic	0.17	0.002	0.041	0.003	0.000	0.001
Mercury	0.9	0.01	0.16	0.016	0.002	0.003
NOx	9259.7	376.4	7038.7	162.5	69.7	143.4
SO2	2294.0	121.3	2191.7	40.2	22.5	44.6
HCl	631.2	25.5	480.3	11.1	4.7	9.8
HF	64.4	3.5	64.8	1.1	0.65	1.32
VOCs	498.1	24.4	358.2	8.7	4.5	7.3
Cl-VOCs	50.7	3.2	60.4	0.89	0.60	1.23
Benzene	1.0E-01	5.0E-03	8.8E-02	1.8E-03	9.2E-04	1.8E-03
Dioxins/Furans	3.4E-04	1.3E-07	2.4E-06	6.0E-06	2.4E-08	4.8E-08
PCBs	1.3E-04	1.4E-05	2.7E-04	2.4E-06	2.6E-06	5.6E-06

Figure 11. Relative contribution of management processes to total emissions in Italy, Slovakia and England.



3.3 Population Exposure to Emissions from Incinerators and Landfills in Italy, Slovakia and England

Table 14 shows the characteristics of the populations living close to the incinerators in the three countries. In Italy, more than a million people are involved, including 9010 newborns; more than 19% of the population is 65+ years and the social class distribution is skewed towards more deprivation (25% in class V (deprived) versus 12.6% in class I (less deprived)). More than half of the population live close to more recent plants built after 1990. The majority of residents within 3 km (64.4%) are located in the 2-3 km circular zone. Only 16,000 people live close to the two incinerators in Slovakia and the age distribution is younger than in Italy. Also in this case, most of the residents live in the longest circle farther away. Contrary to Italy, the social class distribution around the two plants in Slovakia is skewed toward a higher social class. In England, approximately 1,200,000 people live around the 11 incineration plants, mostly in the 2-3 Km circular zone, and the social class distribution is strongly skewed towards deprivation (55% in class V (deprived) versus 3% in class I (less deprived)).

Table 14. Characteristics of residents living within 3 km from an incinerator in Italy, Slovakia and England, 2001.

Variable	Italy		Slovakia		England	
	N	%	N	%	N	%
Total	1060569		16409		1203208	
Sex						
males	511831	48.3	8039	49.0	592817	49.3
females	548738	51.7	8370	51.0	610391	50.7
Age (years)						
0	9010	0.8	176	1.1	16425	1.4
1-14	123061	11.6	2914	17.8	233047	19.4
15-44	435825	41.1	7795	47.5	569850	47.4
45-64	289430	27.3	4337	26.4	229133	19.0
65+	203243	19.2	1187	7.2	154753	12.9
SES						
I	133211	12.6	9127	55.6	35498	3.0
II	159735	15.1	386	2.4	76359	6.3
III	223059	21.0	1600	9.8	150253	12.5
IV	257009	24.2	4856	29.6	274692	22.8
V	264401	24.9	414	2.5	666406	55.4
m.i	23154	2.2	26	0.2		0.0
Period of start						
1960-1970	81586	7.7	0	0.0		0.0
1971-1980	127750	12.0	14240	86.8	533915	44.4
1981-1990	301950	28.5	2169	13.2		0.0
1991-2001	549283	51.8	0	0.0	669293	55.6
Distance						
0-1 Km	50990	4.8	221	1.3	95179	7.9
1-2 km	326798	30.8	3433	20.9	416987	34.7
2- 3 km	682781	64.4	12755	77.7	691042	57.4

Table 15 shows the characteristics of the population living close to landfills in the three countries. The statistics for Italy have been calculated for 118 sites in five regions and then extrapolated to the national level which included 619 sites. In Italy, more than 1,350,000 people are involved, including 11,766 newborns; more than 18% of the population is 65+ years and the social class distribution is skewed towards more deprivation (26% in class V (deprived) versus 13% in class I (less deprived)). The majority of residents within 2 km (85.7%) are located in the 1-2 km circular zone. A total of 328,869 people live close to the 121 landfill plants in Slovakia and the age distribution is younger than in Italy. Also in this case, most of the residents live in the longest circle farther away. In England, a total of 1,425,350 people live close to the 232 geocoded landfills (including 16,242 newborns), especially in the 1-2 Km circular area, and the social class distribution is skewed towards deprivation (20% in class V (deprived) versus 2.5% in class I (less deprived)).

Table 15. Characteristics of residents living within 2 km from landfills in Italy, Slovakia and England 2001.

Variable	Italy observed data*		Italy estimated data**		Slovakia		England	
Total	257513		1350852		328869		1425350	
Sex								
males	125750	48.8	659655	48.8	159822	48.6	694137	48.7
females	131763	51.2	691197	51.2	169047	51.4	731213	51.3
Age								
0	2243	0.9	11766	0.9	3285	1.0	16242	1.1
1-14	32801	12.7	172066	12.7	59450	18.1	260043	18.2
15-44	107244	41.6	562577	41.6	156109	47.5	580430	40.7
45-64	67971	26.4	356560	26.4	76617	23.3	344290	24.2
65+	47254	18.4	247883	18.4	33408	10.2	224345	15.7
SES								
I	34252	13.3	179678	13.3	79591	24.2	35277	2.5
II	38715	15.0	203090	15.0	81172	24.7	254972	17.9
III	57801	22.4	303210	22.4	74349	22.6	266629	18.7
IV	59320	23.0	311179	23.0	53893	16.4	271786	19.1
V	67339	26.1	353244	26.1	39855	12.1	286964	20.1
m.i	86	0.0	451	0.0	9	0.0	309722	21.7
Distance								
0-1 Km	36716	14.3	192603	14.3	59522	18.1	216938	15.2
1-2 km	220797	85.7	1158249	85.7	269347	81.9	1208412	84.8

*Only 118 out of 619 landfills were geocoded for Italy

** estimation is based on data from the 118 landfills

Table 16 shows the results of the application of the local air dispersion model for the three countries. Population-weighted additional exposure to PM_{10} and NO_2 in 2001 is indicated together with standard deviation and percentiles. The estimates for Italy and England derive from models with measured emissions (1st method) or national limits (2n method). The additional contribution to PM_{10} (using the national limit value) is 0.0114 ug/m^3 for Italy, 0.0078 ug/m3 for Slovakia, and 0.0017 ug/m3 for England. The additional contribution to NO2 (using the national limit value) is 0.2271 ug/m3 for Italy, 0.1542 ug/m^3 for Slovakia, and 0.1438 ug/m^3 for England. The use of measured emission values had a strong impact on the estimate for PM_{10} (eg. 0.0030 ug/m3 for Italy) but a lower impact for NO_2 (e.g. 0.1944 ug/m^3 for Italy).

Table 16. Results of the application of the local air dispersion model for PM_{10} and NO_2 around 40 incinerators in Italy, 2 incinerators in Slovakia and 11 incinerators in England. Population-weighted exposure to PM_{10} and NO_2 in 2001 (data in ug/m3).

	Italy	Slovakia	England
PM_{10} (real data)			
Mean (SD)	0.0030 (0.0040)	na	0.0017 (0.0013)
25% percentile	0.0010	na	0.0009
50% percentile	0.0016	na	0.0013
75% percentile	0.0032	na	0.0021
PM_{10} (national limits)			
Mean (SD)	0.0114 (0.0151)	0.0078 (0.0037)	0.0152 (0.0110)
25% percentile	0.0038	0.0066	0.008
50% percentile	0.0061	0.0075	0.0116
75% percentile	0.0120	0.0082	0.0189
NO_2 (real data)			
Mean (SD)	0.1944 (0.2583)	na	0.1438 (0.1272)
25% percentile	0.0658	na	0.0570
50% percentile	0.1050	na	0.1068
75% percentile	0.2060	na	0.1851
NO_2 (national limits)			
Mean (SD)	0.2271 (0.3018)	0.1542(0.0747)	0.3036 (0.2201)
25% percentile	0.0769	0.131	0.1589
50% percentile	0.1220	0.149	0.2329
75% percentile	0.2400	0.163	0.3778

na: data not available

Table 17 shows PM10 and NO2 population-weighted exposure levels by selected characteristics in the three countries. Generally no differences were found for gender and age whereas higher exposure values were found among those of lower socioeconomic status in Italy and England (not in Slovakia). In Italy, the highest exposure values were found for those living around incinerators built between 1981-1990.

Table 17. Population-weighted exposure to PM_{10} and NO_2 in 2001 (based on national limit values) due to incinerators by selected population characteristics (data in ug/m3).

Variable	Italy				Slovakia				England			
	PM_{10}		NO_2		PM_{10}		NO_2		PM_{10}		NO_2	
	mean	**(SD)**	**mean**	**(SD)**	**mean**	**(SD)**	**mean**	**(SD)**	**mean**	**(SD)**	**mean**	**(SD)**
Total	0.0113	(0.0151)	0.2271	(0.3018)	0.0078	(0.0037)	0.1542	(0.0747)	0.0152	(0.0017)	0.3036	(0.2201)
Sex												
males	0.0113	(0.0151)	0.2267	(0.3009)	0.0078	(0.0035)	0.1541	(0.0719)	0.0152	(0.1334)	0.3035	(2.6671)
females	0.0113	(0.0149)	0.2264	(0.2974)	0.0077	(0.0037)	0.1533	(0.0744)	0.0152	(0.1334)	0.3039	(2.6672)
Age												
0	0.0111	(0.0150)	0.2214	(0.2990)	0.0074	(0.0013)	0.1480	(0.0271)	0.0156	(0.0110)	0.3039	(0.2197)
1-14	0.0114	(0.0154)	0.2276	(0.3079)	0.0080	(0.0036)	0.1589	(0.0718)	0.0156	(0.0112)	0.3120	(2.6672)
15-44	0.0113	(0.0150)	0.2253	(0.3006)	0.0079	(0.0037)	0.1567	(0.0747)	0.0149	(0.0109)	0.2985	(0.2172)
45-64	0.0113	(0.0148)	0.2264	(0.2950)	0.0074	(0.0032)	0.1463	(0.0650)	0.0152	(0.0109)	0.3047	(0.2187)
65+	0.0113	(0.0143)	0.2255	(0.2853)	0.0065	(0.0021)	0.1271	(0.0451)	0.0154	(0.0113)	0.3086	(0.2263)
SES												
I	0.0108	(0.0132)	0.2162	(0.2648)	0.0070	(0.0019)	0.1383	(0.0409)	0.0111	(0.0074)	0.2227	(0.1477)
II	0.0113	(0.0166)	0.2252	(0.3308)	0.0072	(0.0015)	0.1254	(0.0267)	0.0136	(0.0084)	0.2716	(0.1687)
III	0.0117	(0.0151)	0.2340	(0.3019)	0.0126	(0.0077)	0.2519	(0.1546)	0.0157	(0.0115)	0.3145	(0.2300)
IV	0.0112	(0.0143)	0.2231	(0.2862)	0.0078	(0.0034)	0.1564	(0.0682)	0.0153	(0.0120)	0.3055	(0.2402)
V	0.0133	(0.0171)	0.2652	(0.3423)	0.0079	(0.0009)	0.1583	(0.0179)	0.0154	(0.0120)	0.3084	(0.2165)
m.i	0.0057	(0.0060)	0.1136	(0.1196)								
Period of start												
1960-1970	0.0106	(0.0104)	0.2113	(0.2078)								
1971-1980	0.0095	(0.0145)	0.1893	(0.2910)	0.0083	(0.0035)	0.1667	(0.0707)	0.0157	(0.0112)	0.3132	(0.2238)
1981-1990	0.0250	(0.0250)	0.4990	(0.4991)	0.0039	(0.0018)	0.0675	(0.0315)				
1991-2001	0.0106	(0.0133)	0.2114	(0.2654)					0.0148	(0.0109)	0.296	(0.2173)

3.4 Quantification of Health Effects due to incinerators

Table 18 and figures 12-14 show the estimated number of additional cancer incident cases in the three countries for the period 2001-2050 as a result of exposure before 2001 (past exposure) and during 2001-2020 (current exposure). In Italy, 7300 cases are expected annually in the population living close to incinerators. An additional number of approximately 90 cases per year will be attributable to past exposure up to 2020 and then the number will decline to a minimum of 1.6 in 2050. On the other hand, the annual number of cases due to current exposure increases to 11 in 2020 and then will decline to 0 in 2050. In total, 2729 (95%CI=2334-3112) additional cancer cases will be attributable to incinerators in Italy during 2001-2050 and the vast majority of them are due to exposure before 2001. The total number of cancers attributable to exposure during 2001-2020 is 189 (95%CI=162-216).

In Slovakia, 46 cancer cases are expected annually in the population living close to the two incinerators. Less than one additional case per year is estimated for past exposure during the whole period whereas the estimate for current exposure is very low. In total, 24 (95%CI=21-28) additional cancer cases will be attributable to incinerators in Slovakia during 2001-2050 and the majority of them are due to exposure before 2001. The total number of cancers attributable to exposure during 2001-2020 is 1.2 (95%CI=1.0-1.4).

In England, 3200 cancer cases are expected annually in the population living close to incinerators. An additional number of approximately 36 cases per year will be attributable to past exposure up to 2020 and then the number will decline to 0 in 2050. On the other hand, the annual number of cases due to current exposure increases to 7 in 2020 and then will decline to 0 in 2050. In total, 1125 (95%CI=964-1286) additional cancer cases will be attributable to incinerators in England during 2001-2050 and the vast majority of them are due to exposure before 2001. The total number of cancers attributable to exposure during 2001-2020 is 120 (95%CI=103-137).

Table 18. Estimated number of additional cancer cases in the three countries as result of exposure to incinerators before 2001 (past exposure) and during 2001-2020 (current exposure).

	ITALY		SLOVAKIA		England	
	Additional cases	95% CI	Additional cases	95% CI	Additional cases	95% CI
Cases due to exposure before 2001 (Past exposure)						
2001	88	76 - 101	0.82	0.71 - 0.94	33	28 - 38
2010	92	79 - 105	0.85	0.73 - 0.98	36	31 - 41
2020	89	76 - 101	0.84	0.72 - 0.96	36	31 - 41
2030	28	24 - 32	0.28	0.24 - 0.32	12	10 - 13
2040	2.0	1.4 - 2.6	0.002	0.001 - 0.002	0.07	0,06 - 0,08
2050	1.6	1.1 - 2.1	0	0 - 0	0	0 - 0
Total	*2540*	*2172-2896*	*23*	*20 - 27*	*1005*	*861-1149*
Cases due to exposure during 2001-2020 (Current exposure)						
2001	0	0 - 0	0	0 - 0	0	0 - 0
2010	2.7	2.3 - 3.1	0.017	0.015 - 0.020	1.7	1,5 - 2,0
2020	11	10 - 13	0.071	0.061 - 0.081	7.1	6,1 - 8,1
2030	4.6	4.0 - 5.3	0.029	0.025 - 0.033	2.9	2,5 - 3,3
2040	0.05	0.04 - 0.06	0	0 - 0	0.0	0 - 0
2050	0	0 - 0	0	0 - 0	0.0	0 - 0
Total	*189*	*162-216*	*1.2*	*1.0 - 1.4*	*120*	*103 - 137*
Total (Past + Current exposure)						
2001	88	76 - 101	0.82	0.71 - 0.94	33	28 - 38
2010	95	81 - 108	0.87	0.75 - 1.0	38	33 - 43
2020	100	86 - 114	0.91	0.78 - 1.0	43	37 - 49
2030	33	28 - 37	0.31	0.026 - 0.035	15	13 - 16
2040	2.1	1.4 - 2.7	0.002	0.001 - 0.002	0.1	0,09 - 0,12
2050	1.6	1.1 - 2.1	0	0 - 0	0.0	0 - 0
Total	*2729*	*2334-3112*	*24*	*21 - 28*	*1125*	*964-1286*

Figure 12. Annual number of additional cancer cases near incinerators in <u>Italy</u> attributable to exposure before 2001 (past exposure) and during 2001-2020 (current exposure).

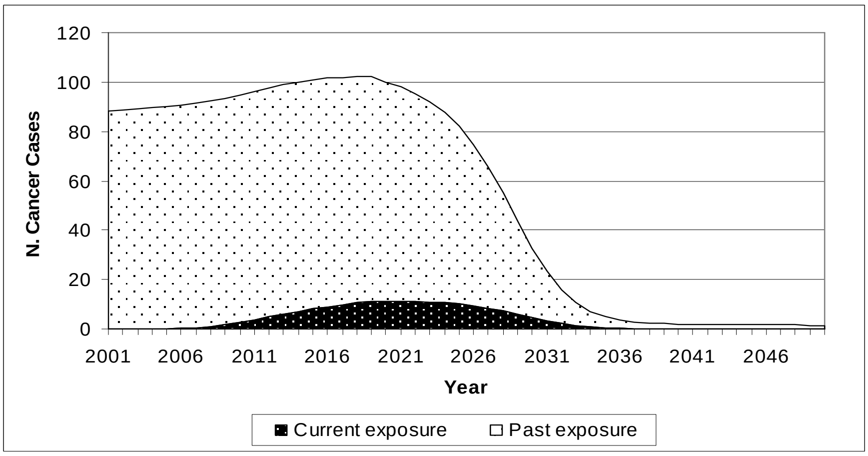


Figure 13 Annual number of additional cancer cases near incinerators in <u>Slovakia</u> attributable to exposure before 2001 (past exposure) and during 2001-2020 (current exposure).

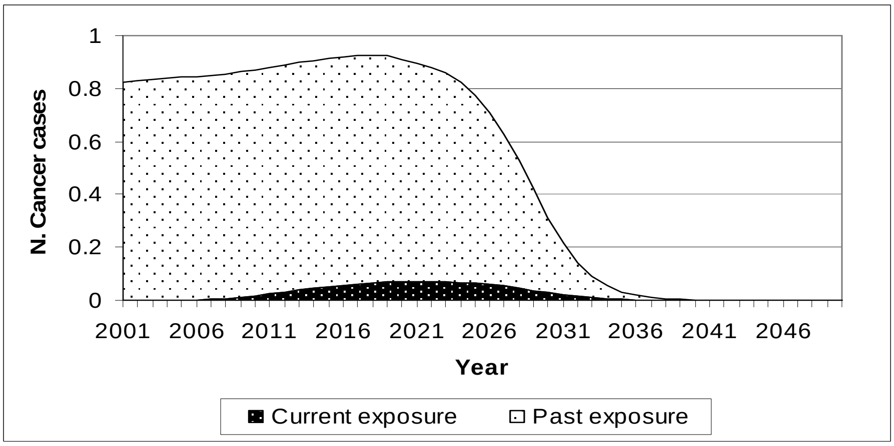
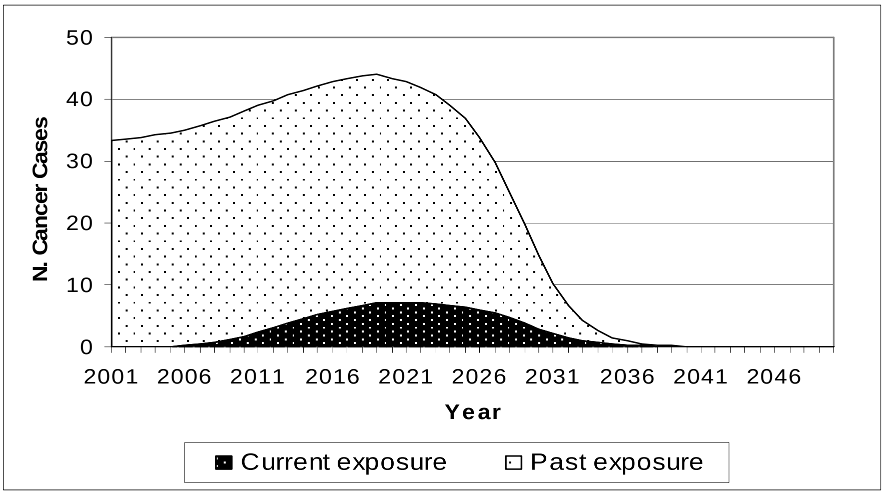


Figure 14 Annual number of additional cancer cases near incinerators in <u>England</u> attributable to exposure before 2001 (past exposure) and during 2001-2020 (current exposure).



In order to estimate the impact of PM10 or NO2 additional exposures on life expectancy, we applied the coefficient of increased mortality per 10 ug/m3 (0.06 for both pollutants) to the estimated increase in the pollution level to the exposed population in each country. For instance, the calculations of the excess risk of mortality for NO2 (0.2271 ug/m3) in Italy were:

0.06/10 * 0.2271=0.0013626

The value has been applied to the existing exposed population for the years 2001-2020. Estimates were made with a follow up to 2050. Table 19 shows the total number of Years of Life Lost, YoLL (also the YoLL per 100,000 inhabitants and the number of lost days per person) in the three countries attributable to exposure to PM10 or NO2 from incinerators. In Italy, the impact is higher for NO_2 (total YoLL 3603, 341.4 per 100,000 inhabitants) than for PM_{10} (total YoLL 181, 17.16 per 100,000 inhabitants). In Slovakia, the total number of YoLL is also higher for NO_2 (37, 225 per 100,000) than for PM_{10} (2, 12.2 per 100,000). Comparable results were available for England with a total impact similar to Italy (for NO_2: total YoLL 4127, 350 per 100,000 inhabitants; for PM10: total YoLL 211, 17.5 per 100,000 inhabitants). Overall, the maximum impact of incinerators is 1.25 days per each person in Italy, 0.82 days per person in Slovakia, and 1.28 days per person in England.

Table 19. Estimated number of Years of Life Lost (follow up to 2050) in the three countries as result of exposure to PM_{10} and NO_2 from incinerators.

	ITALY			SLOVAKIA			ENGLAND		
	Total YLL	**YLL / 100000**	**days / person**	**Total YLL**	**YLL / 100000**	**days / person**	**Total YLL**	**YLL / 100000**	**days / person**
PM10 (real data)	5	0.47	0.002	n.a	n.a	n.a	23	1.91	0.007
PM10 (national limits)	181	17.16	0.06	2	12.19	0.04	211	17.54	0.06
NO2 (real data)	3084	292.2	1.07	n.a	n.a	n.a	1999	166.1	0.61
NO2 (national limits)	3603	341.4	1.25	37	225.5	0.82	4217	350.5	1.28

n.a.: not available

When the effects of emissions from incineration were evaluated for Europe using large scale dispersion models for the emissions of NH3, NMVOC, NOX, PPMcoarse, PPM2.5 and SO2, the estimated YoLL per year were 38.6 (real emission values) or 56.4 (national limits) in Italy, 2.45 (for exposure after 2002) in Slovakia, and 24.1 in England for the plants considered.

3.5 Quantification of Health Effects due to landfills

Table 20 shows the health effects of landfills in the three countries as annual cases of congenital malformations and newborns of low birth weight during the period 2001-2030. It is expected that an average of 1.96 (95%CI=0.98-2.94) additional cases of birth defects per year occur in Italy, 1.54 (0.77-2.31) in Slovakia and 2.7(1.35-4.0) in England. The estimated number of infants of low birth weight is 42 (95%CI=35-42), 13 (11-13), and 58 (49-58) cases per year for 30 years, respectively for the three countries.

Table 20. Estimated health effects of exposures to landfills in the three countries as annual cases of congenital malformations and newborns of low birth weight.

	Italy			Slovakia			England		
	Expected cases	Additional cases	99% CI	Expected cases	Additional cases	99% CI	Expected cases	Additional cases	99% CI
All congenital anomalies	73	1.47	0.73 - 2.20	77	1.54	0.77 - 2.31	83	2.7	1.35 - 4.05
Neural tube defects	6	0.37	0.06 - 0.74	2	0.11	0.02 - 0.23	5	0.31	0.05 - 0.62
Hypospadias and epispadias	10	0.67	0.38 - 1.06	7	0.48	0.27 - 0.75	16	1.13	0.65 - 1.78
Abdominal wall defects	2	0.08	0-0.33	12	0.60	-0.72 - 1.92	5	0.27	-0.32 - 0.86
Gastroschisis and exomphalos	2	0.27	0.05 - 0.51	12	2.16	0.36 - 4.09	5	0.85	0.14 - 1.61
Low birth weight	706	42.4	35.3-42.4	212	12.7	10.62 - 12.74	975	58.5	48.7 - 58.5

3.6 Sensitivity analysis

All planned sensitivity analyses will be conducted during the 2nd pass assessment.

4. Discussion

4.1 Main findings

We found that the amount of MSW produced in Italy, Slovakia and England are comparable to what has been estimated for the entire EU (EU-15: 570 kg per inhabitant; EU-12:335 kg per inhabitant, 2004 data) (EEA, 2008). In 2001, the three countries differed with regard to recycling, landfilling and incineration policies: Slovakia and England were the countries where landfills were the most important method of management whereas Italy had the highest proportion of recycling and use of MBT technologies; incineration was used equally in Italy and England. This diversity has been observed and already well described in Europe (EEA, 2007). Despite large differences in management, the amount of emitted air pollutants per inhabitant is roughly similar in the three countries although estimated metals and dioxins emissions in Italy are higher than in the other two nations because of larger emissions of these pollutants from Italian incinerators. There is a sizeable population living close to management plants in the three areas (e.g. approximately 2% of the entire population in Italy live close to an incinerator while an additional 2.5% live close to a landfill). In both Italy and England, populations with lower socio-economic status are prone to live closer to waste disposal sites. Since lower socio-economic status is already associated with a higher risk of various negative health outcomes, an issue of environmental justice is present here because of the higher probability of exposure for less affluent people and their increased vulnerability. The situation is different for the two incinerators in Slovakia since they have an urban location and people living in urban areas in that country tend to have a higher socioeconomic profile. Confirming preliminary research in the UK (Mindell & Barrowcliffe, 2005), the additional contribution to the PM_{10} and NO_2 background in proximity of incinerators estimated with air dispersion models is relatively small and roughly equivalent in the three countries. After a systematic review of the scientific literature, we found that cancer incidence and adverse reproductive outcomes (congenital malformations and low birth weight) are the main health effects possibly related to incinerators and landfills, respectively. These findings emphasize the need to consider two subgroups of the population as particularly vulnerable to possible negative health effects of waste management and waste treatment practices – pregnant women, or women of reproductive age in general, and the elderly population.

On the basis of the excess risk derived from published data, we found that the largest health impact from incinerators during the period of evaluation (2001-2050) is cancer incidence accounting for a small percentage increase over the background in the exposed population. The majority of the cancer cases are due to exposures occurring before 2001 whereas the relative impact from the current exposure pattern is less important. It is therefore important to consider the health burden that is not amenable to intervention from new policies since those cancer cases will occur in any case. On the other hand, policies for future developments should consider that most of the health effects will be seen over several decades. The application of the air dispersion model data to a life table analysis indicates that the maximum impact of incinerators on the overall mortality of the resident cohort will be from exposure to NO_2. A few hundred of Years of Life Lost per 100,000 people over the period 2001-2020 are expected to occur and the results are surprisingly consistent over the three countries. However, the burden estimated with a large scale model for the entire European population should be added to the overall impact of incineration as the impact is widespread.

Our evaluation of the impact of landfills is driven from the relative lack of scientific knowledge related to health effects since only adverse reproductive disorders were considered. The overall estimated burden in each country consists of few cases of congenital malformations and low birth weight newborns. It is likely that most of the overall impact of landfills comes from greenhouse gasses and their direct influence on climate change.

4.2 Comparison with other assessments

In our project, the study population comprises the general populations of Italy, Slovakia and England. The local target population was defined on the basis of their distance from landfills and incinerators. In the case of incinerators, estimates of pollutant concentrations based on local and large scale dispersion modelling was used to define the target population. It is very difficult then to compare our results with other assessments.

There are several examples in the literature of risk assessment of a single or a limited number of waste management plants (e.g. Mindell & Barrowcliffe, 2005). Results of risk assessment performed at the country level are more limited, although the ExternE methodology (Rabl and Spadaro, 2002; Rabl et al. 2008) has been applied to estimate external costs of waste management. Rabl et al. (2008) concluded that the only significant contributions come from direct emissions (of the landfill or incinerator) and from avoided emissions due to energy recovery (from an incinerator). Damage costs for incineration range from about 4 to 21 EUR tonne waste, and they are extremely dependent on the assumed scenario for energy recovery. For landfills the costs range from about 10 to 13 EUR tonne waste; it is dominated by greenhouse gas emissions because only a fraction of the CH4 can be captured. A complete assessment has been conducted in Singapore (Tan & Khoo, 2006) but with the main focus on environmental impact. Experiences of the health impact assessment in Europe are available from Ireland (Health Research Board Ireland, 2003) and England (Enviros, 2004). The latest study provides a wide review focused on the environmental and health effects of MSW management. The study was published by DEFRA (Department for Environment, Food and Rural Affairs of England) and performed by Enviros Consulting, Ltd. in cooperation with the University of Birmingham. The methods this study used have been relevant to the present health impact assessment.

4.3 Limitations of the assessment

Our health impact assessment is characterised by a number of uncertainties that are typical of these exercises when applied to long term-effects of prolonged, low-level exposures, or exposures occurring at critical stages of development (e.g. childhood or pre-natal exposure). We have listed the sources of uncertainties for each step of our evaluation and briefly summarize our confidence in the methods and results.

1. Waste generation and management

As expected, there were inadequacies in data availability and reliability on MSW indicators as they are not uniform and not always available in the same format from published statistics. There were approximations in the available information on waste composition, classification of wastes is different in different countries, and we had high uncertainty concerning the amount and treatment of illegally disposed wastes. Overall, however, we have high confidence in the summary statistics reported.

2. Emissions of pollutants from waste management facilities

We estimated total emissions from waste management facilities using the amount of managed waste and tabulated emission factors from the literature. These emission factors have a wide range of uncertainties, some of which have been evaluated and can be quantified (Enviros, 2004). It should be noted, however, that measured emissions data are sparse, several pollutants are not measured, and data on some approaches to waste management (composting, gasification, illegal disposal) are difficult to find. Finally, the emission factors that we considered are based on facilities under normal operational activity and there is the possibility of accidental releases that are difficult to quantify. Overall, we have high confidence in our emissions estimates from incinerators whereas we have moderate confidence in the other technologies, including transportation.

3. Population characteristics and exposure to air pollutants

While we had relatively high quality data for incinerators in the three countries, exact coordinates of landfills were difficult to find in Italy. In addition, we did face difficulties in estimating the exposed population because the location of the plant was approximate, the size of some landfills is not known, and the unit of the available population data (census block) does not fit our needs. We were fortunate because the population data by age and sex is available at the local level, however they are based on the census and approximations were made for years beyond the census. Overall, we have very high confidence on the population data close to incinerators but our confidence on population data close to landfills is moderate.

The results of the air dispersion models depend on the quality of the data. We had operational data measured during recent years for some of the incinerators but only estimated emissions for some others. In addition, some plant characteristics were missing and had to be inputted. On the other hand, we could rely on high quality meteorological data for most of the plants and topography was also considered. Overall, we have a high confidence in the estimated air pollution concentrations close to incineration plants.

4. Excess-risk and exposure-response functions

The application of excess-risk estimates based on distance from the plants has been problematic because of several difficulties in interpreting epidemiological studies. We have tried to address the issue in a transparent way by conducting a systematic evaluation, however, as underlined on several occasions, we have moderate confidence in the excess risks used for the impact assessment of cancer cases and adverse reproductive outcomes. On the other hand, we have high confidence in the coefficients for long-term effects of PM_{10} and NO_2 on mortality.

5. Quantification of the health impact.

The quantification has been straightforward in terms of calculating excess cases as there are no difficulties in finding the appropriate health statistics and in taking into account the particular population characteristics near the facilities. However, the most difficult part is attributing the effect studied from old plants using old technologies to new facilities. We have clearly stated our assumptions and also have tried to evaluate the consequence of changing some of the parameters. Overall, we have moderate confidence in our method to estimate excess cancer cases and reproductive outcomes. On the other hand, the life table approach is rather robust although it is difficult to verify some of the assumptions (time of the effect, stability of the population, constant mortality). Finally, because a variety of illegal disposal practices exist and because it is difficult to estimate the amount of waste that is disposed of illegally, determining emissions, exposure levels and health effects is difficult. For all of these reasons, our quantification of the health impacts has a moderate level of confidence.

5. Conclusions

The main results of the present study should be viewed in relation to the present debate within the EU on the main policy issues related to waste management. Open questions that remain are the effort that individual countries should make to reduce the overall amount of waste, and the appropriate targets to be meet for recycling. Although waste to energy is gradually replacing old mass incineration, open questions remain over the extent to which such policies should be introduced. There are several uncertainties and critical assumptions in our assessment model that are typical of a complex problem. However, we believe that it provides insight into the relative health impact attributable to waste and that the model could potentially be useful for evaluating future proposed policies.

References

Briggs D, de Hoogh C, Hurt C and Maitland I. 2002 *A geographical analysis of populations living around landfill sites. London: Small Area Health Statistics Unit, Imperial College*. SAHSU Technical Report. (http://www.sahsu.org/pdf/techreportv6.pdf)

Cadum E, Costa G, Biggeri A, Martuzzi M. 1999 [Deprivation and mortality: a deprivation index suitable for geographical analysis of inequalities] *Epidemiologia & Prevenzione* 23:175-87. Italian.

CERC Ltd, 1999, ADMS-Urban. *An urban air quality management system*. User guide. Cambridge: CERC Ltd.

Comba P, Ascoli V, Belli S, Benedetti M, Gatti L, Ricci P, Tieghi A. 2003 Risk of soft tissue sarcomas and residence in the neighbourhood of an incinerator of industrial wastes. *Occupational & Environmental Medicine*. 60:680-3.

Cordier S, Chevrier C, Robert-Gnansia E, Lorente C, Brula P, Hours M. 2004 Risk of congenital anomalies in the vicinity of municipal solid waste incinerators. *Occupational & Environmental Medicine*. 61:8-15.

Crowley. 2003 Health Research Board Ireland. *Health and Environmental Effects of Landfilling and IIncineration of Waste – A Literature Review*. Health Research Board. www.hrb.ie

Preiss, Ph. and Klotz V. *2008 EcoSenseWeb, User's Manual. http://ecosenseweb.ier.uni-stuttgart.de*

ENEA. 2006 Rapporto sul recupero energetico dei rifiuti urbani in Italia. Enea, Roma.

Elliott P, Briggs D, Morris S, de Hoogh C, Hurt C, Kold Jensen T, Maitland I, Richardson S, Wakefield J, Jarup L. 2001 *Risk of adverse birth outcomes in populations living near landfill sites. British Medical Journal* 323:363–8.

European Environmental Agency. 2007 *The road from landfilling to recycling: common destination, different routes*. EEA, Copehnagen. http://reports.eea.europa.eu/brochure_2007_4/en/Landfill_brochure.pdf

European Environmental Agency. 2008 *Better management of municipal waste will reduce greenhouse gas emissions*. EEA, Copehnagen.

Enviros, 2004 *Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes*. Department of Environment Transport and the Regions (DEFRA), UK.

European Commission. 2005 *Taking sustainable use of resources forward: A Thematic strategy on the prevention and recycling of waste*. Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the regions, 32 pp.

ExternE-Project series

- European Commission, DG XII, Science, Research and Development, JOULE. 1999 *ExternE - Externalities of Energy*. Volume 7: Methodology 1998 update, EUR 19083.
- European Commission. 2005 *ExternE - Externalities of Energy* – Methodology 2005 Update. EUR 21951 EN. Office for Official Publications of the European Communities, Luxembourg. ISBN 92-79-00423-9.

Available at: www.ExternE.info

Floret N, Mauny F, Challier B, Arveux P, Cahn JY, Viel JF. 2003 Dioxin emissions from a solid waste incinerator and risk of non-Hodgkin lymphoma. *Epidemiology*. 14:392-8.

Gilbreath S, Kass PH. 2006 Adverse birth outcomes associated with open dumpsites in Alaska Native Villages. *American Journal Epidemiology*. 164:518-28.

Goldberg MS, Siemiatyck J, DeWar R, Desy M, Riberdy H. 1999 Risks of developing cancer relative to living near a municipal solid waste landfill site in Montreal, Quebec, Canada. *Archives of Environmental Health*. 54:291-6.

Intergovernmental Panel on Climate Change. 2005 *Guidance notes for lead authors on the IPCC fourth assessment report on adressing uncertainty*. http://ipcc-wg1.ucar.edu/wg1/Report/AR4_UncertaintyGuidanceNote.pdf. 2005.

Jarup L, Briggs D, de Hoogh C, Morris S, Hurt C, Lewin A, Maitland I, Richardson S, Wakefield J, Elliott P. 2002 Cancer risks in populations living near landfill sites in Great Britain. *British Journal of Cancer* 86:1732-1736.

Mindell J, Barrowcliffe R. 2005 Linking environmental effects to health impacts: a computer modelling approach for air pollution. *Journal of Epidemiology & Community Health*. 59:1092-8.

Morselli L, Luzi J, De Robertis C, Vassura I, Carrillo V, Passarini F. 2007 Assessment and comparison of the environmental performances of a regional incinerator network. *Waste Management* 27:S85-91.

Morselli L, De Robertis C, Luzi J, Passarini F, Vassura I. 2008 Environmental impacts of waste incineration in a regional system (Emilia Romagna,Italy) evaluated from a life cycle perspective. *Journal of Hazardous Materials*. 23. [Epub ahead of print]

National Society for Clean Air. 2002 *Comparison of emissions from waste management options*. http://www.nsca.org.uk/assets/EFWEA%20full%20report.pdf

National Society for Clean Air. 2002 *Relative impacts of transport emissions in recycling*. http://www.nsca.org.uk/assets/EAPP%20full%20report.pdf

NEEDS 2008.European Commission (EC) – Integrated project NEEDS – New Energy Externalities Developments for Sustainability, Project no: 502687, Research Stream 1b. *Development and improvement of a methodology to estimate external costs of energy*. - http://www.needs-project.org

Pukkala E, Ponka A. 2001 Increased incidence of cancer and asthma in houses built on a former dump area. *Environmental Health Perspectives*. 109:1121-5.

Rabl A and Spadaro JV. 2002 *Health impact of waste incineration*. In "Environmental and Health Impact of Solid Waste Management Activities. The Royal Society of Chemistry.

Rabl A, Spadaro JV, Zoughaib A. 2008 Environmental impacts and costs of solid waste: a comparison of landfill and incineration. *Waste Management & Research*. 26:147-62.

SEDAC, 2006. Gridded Population of the World, version 3 (GPWv3), Center for International Earth Science Information Network (CIESIN), Columbia University; and Centro Internacional de Agricultura Tropical (CIAT). 2005. Gridded Population of the World Version 3 (GPWv3). Palisades, NY: Socioeconomic Data and Applications Center (SEDAC), Columbia University. http://sedac.ciesin.columbia.edu/gpw

Spadaro JV and Rabl A, 2007. NEEDS project, FP6, Rs1b_D7.2 *Report on the methodology for the consideration of uncertainties*. ARMINES/Ecole des Mines, 60 boul. St.-Michel, 75272 Paris 06, Joseph V. Spadaro, spadarojv@aol.com and Ari Rabl, ari.rabl@ensmp.fr.

Strange K. 2002 *Overview of the waste management options with a discussion of their efficacy and acceptability*. In "Environmental and Health Impact of Solid Waste Management Activities. The Royal Society of Chemistry.

Tan RBH, Khoo HH. 2006 Impact assessment of waste management options in Singapore. *Journal of Air Waste Management Association* 56:244-254.

Torfs R., Hurley F. and Miller B. 2007 NEEDS project, FP6, Rs1b_D3.7. *Final Report on casual links between pollutants and health impacts – A set of concentration-response functions*.

Tarrasón, L. 2008 NEEDS project, FP6, Rs1b_TP1.2. *Documentation and evaluation of the source-receptor calculations made for NEEDS*. Norwegian Meteorological Institute – Research and Development Department – Section Air Pollution, in progress.

Viel JF, Arveux P, Baverel J, Cahn JY. 2000 Soft-tissue sarcoma and non-Hodgkin's lymphoma clusters around a municipal solid waste incinerator with high dioxin emission levels. *American Journal of Epidemiology*. 152:13-9.

World Health Organization (WHO). 1996 *Waste Incineration.* Environmental Health planning pamphlet series. World Health Organisation Regional Office for Europe: Copenhagen.

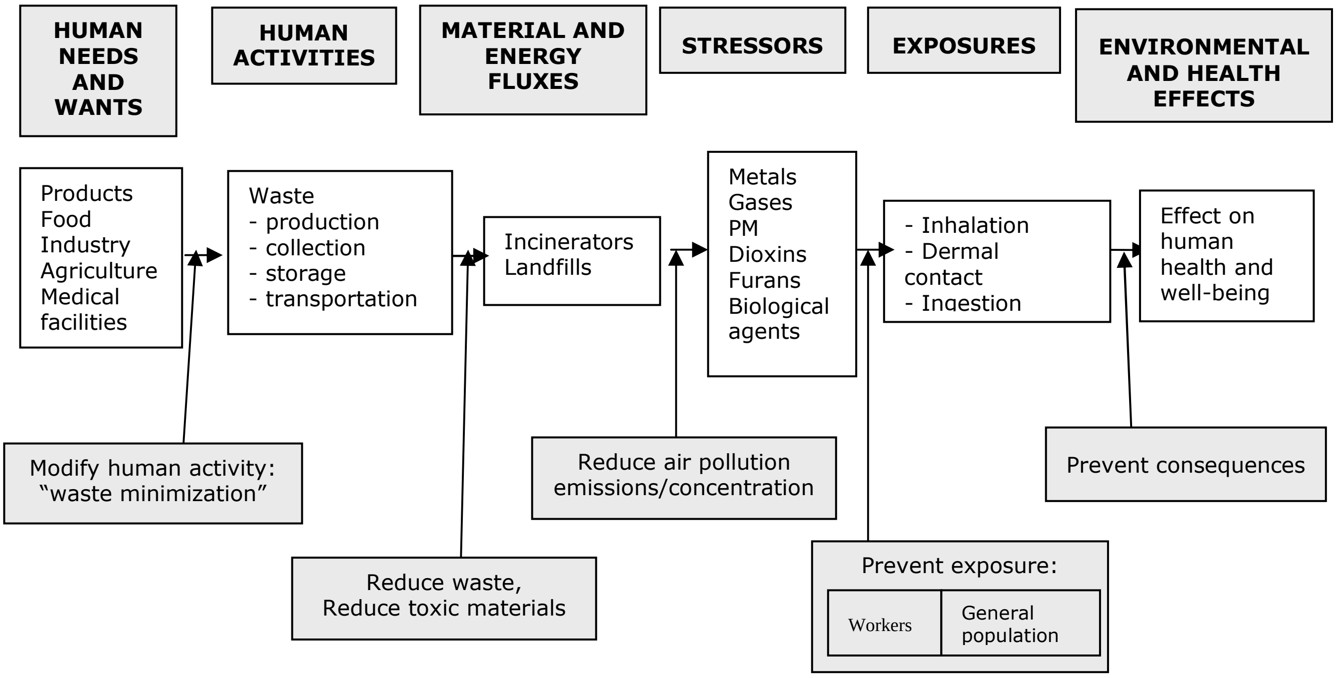
World Health Organization (WHO). 2007 Population health and waste management: scientific data and policy options. Report of a WHO workshop Rome, Italy, 29–30 March 2007. WHO, Regional Office for Europe, Copenhagen.

Zambon P, Ricci P, Bovo E, Casula A, Gattolin M, Fiore AR, Chiosi F, Guzzinati S. 2007 Sarcoma risk and dioxin emissions from incinerators and industrial plants: a population-based case-control study (Italy). *Environmental Health*. 6:19.

APPENDIX 1

Waste production, management and related environmental pressures

Figure 1: The waste management flow and possible policy actions



- Durable, long-lasting goods; - Products and packaging as free of toxins as possible; - Products with fewer raw materials in production, have a longer life, or be used again after its original use.	- Reuse, - Recycling - Buying recycled products - Composting - Separate domestic and hazardous waste - avoid mixing industrial and municipal waste	- Gas temperature reduction techniques - Air pollution control techniques - Environmental monitoring -Ground (and superficial) water contamination control techniques	- Consider regional scale - Consider multiple media -Individual protection equipment	No options

An overview of the complex problem of waste production/disposal, human behaviours on waste, together with actions to be taken in order to prevent adverse effects from waste disposal are showed in Figure 1. The diagram starts with the main driving forces associated with waste production, namely products and goods to fulfill human needs, commercial and industrial activities. As a result, waste is produced with subsequent necessity to collect/store/transport and finally dispose of it. The waste disposal methods currently used represent the source of a wide range of environmental pollutants with, following human exposure, possible deleterious effects on the health of the population. At every level of this process, from waste production to public health and environmental health issues as the end consequences, the diagram points out steps where actions could be taken, or human behaviour could be modified, with the final aim to prevent, or at least minimize, waste production and decrease negative impact of waste treatment / management on environment and human health.

There is a long list of stakeholders within the waste management process including industry, central / regional governments, city councils, NGOs, service users, private companies dealing with waste, citizens, scientists, and the media. It should be noted that several stakeholders are present in the waste management area especially before waste formation: industry, packing, delivery of goods, and citizens are all involved in waste formation as well as in the "waste minimization" process. On the other hand, there are several stakeholders at the end of the process where "wastes" represent important economical resources of material (glass, paper, etc) and energy. Since environmental control is also crucial at the end of the process, public institutions play an important role. There are several conflicting interests among the various stakeholders, e.g. national policy versus local policy, industrial interests versus environmental interests, environmental sustainability and employment, waste minimization and energy production. These conflicting interests, together with citizens' concerns of health effects, make the choices confronting the management of waste very controversial.

Municipal solid waste (MSW) includes predominantly household waste with sometimes the addition of commercial wastes collected by a municipality within a given area. Of course, industry, agriculture and medical facilities produce large quantities of waste, but their direct contribution to MSW is low. Municipal solid wastes are in either solid or semisolid form and can be classified as biodegradable waste (food & kitchen waste, green waste), recyclable material (paper, glass, bottles, cans, metals, certain plastics, etc), inert waste (construction and demolition waste etc), composite wastes (waste clothing, tetra packs, waste plastics), domestic hazardous waste (medication, paints, chemicals, light bulbs, fluorescent tubes, spray cans, fertilizer and pesticide containers, batteries, shoe polish).

The following processes represent the predominant waste management technologies (Enviros, 2004), each being the source of a number of environmental emissions:

1. Materials recycling facilities (MRF)

Materials recycling facility (materials recovery facility) is defined as a central operational plant where source segregated, dry recyclable materials are sorted mechanically or manually for processing into secondary materials. Waste material entering an MRF has normally been subject to some pre-segregation, but further sorting is required which may involve machinery or may involve human contact.

The greatest hazard is related to biological materials, particularly bioaerosol. The associated risks are very similar to those occurring in a composting plant (see below), 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 the plant.

The environmental pressures related to recycling can be summarized as follows:

Air: emission of dust and bioaeresols

Soil: landfilling of final residues

2. Composting

Waste materials that are organic in nature (plant material, food scraps, and paper products) are put through a composting and/or digestion system to control the biological process to decompose the organic matter and kill pathogens. The resulting stabilized organic material is then recycled for agricultural or landscaping purposes. There are a large variety of composting and digestion methods and technologies, varying in complexity from simple window composting of shredded plant material, to automated enclosed-vessel digestion of mixed domestic waste. These methods of biological decomposition are differentiated as being aerobic (composting methods) or anaerobic (digestion methods).

When composting materials are moved, the formation of greenhouse gases and bioaerosol is an inevitable consequence. During optimal management, the composting process generates temperatures sufficient to destroy most pathogenic bacteria. However, these may still survive in any part of the compost that does not reach an adequate temperature and can also be subject to aerosolisation (i.e. becoming suspended in the air). Specific components of the bioaerosol generated during composting are Fungi, Bacteria, Actinomycetes, Endotoxin, Mycotoxins, Glucans.

The environmental pressures related to composting can be summarized as follows:

Air:	emissions of methane (CH_4), carbon monoxide (CO) and bioaerosol

3. Mechanical and biological treatment

Mechanical biological treatment (MBT) is a technology for combinations of mechanical sorting and biological treatment of organic municipal waste. The "mechanical" element is usually a bulk handling mechanical sorting stage. This either removes recyclable elements from a mixed waste stream (such as metals, plastics and glass) or processes it in a given way to produce a high calorific fuel called refuse derived fuel (RDF) that can be incinerated or used in cement kilns or power plants. The "biological" element refers to either anaerobic digestion or composting. Anaerobic digestion breaks down the biodegradable component of the waste to produce biogas and soil conditioner. The biogas can be used to generate renewable energy.

4. Pyrolysis/Gasification with energy recovery

Pyrolysis and gasification are two related forms of thermal treatment where waste materials are heated to high temperatures with limited oxygen availability. The process typically occurs in a sealed vessel under high pressure. Converting material to energy this way is more efficient than direct incineration, with more energy able to be recovered and used. Pyrolysis is the thermal degradation of waste in the absence of air to produce gas (often called syngas), liquid (pyrolysis oil) or solid (char, mainly ash and carbon). The solid components may be subsequently fed into a gasification process. Gasification takes place at higher temperatures than pyrolysis with a controlled amount of oxygen. The majority of the carbon content in the waste is converted into a gaseous form (syngas). The gas produced contains toxic substances similar to those emitted from incinerators.

The environmental pressures related to gasification/pyrolysis can be summarized as follows:

Air:	emission of particulate matter (PM), SO_2, NOx, HCL, HF, non-methanic volatile organic compounds (NMVOC), CO, CO_2, N_2O, dioxins, furans, heavy metals (Zn, Pb, Cu, As, Ni, Hg, Cd)
Water:	deposition of hazardous substances on surface water
Soil:	landfilling of ashes
Ecosystem:	contamination and accumulation of toxic substances in the food chain

5. Incineration (with and without energy recovery)

Incineration is a waste disposal method that involves the combustion of waste at high temperatures ("thermal treatment"). Incineration of waste materials converts the waste into heat, gaseous emissions, and residual solid ash. There are three main approaches that have been adapted for the incineration of municipal waste: mass burning, fluidised bed and refuse derived fuel (RDF) (WHO 1996).

Mass burning refers to the combustion of municipal waste with only rudimentary preparation and separation of the waste. A variety of moving grates have been used to facilitate the movement of the waste through the combustion zone. The grate ensures the passage of the burning refuse through the combustion zone and also allows the provision of adequate supplies of air to guarantee complete combustion of the waste, and ash removal.

In fluidised bed systems, smaller combustion units are used and there is some pre-processing of the waste to facilitate the operation of the fluidised bed. A bed of inert solid particles is fluidised by the flow of combustion air from beneath the bed. Pre-treatment of the waste provides a uniform feed material. In RDF systems, municipal waste is pre-processed using several sorting and shredding stages to produce a stable dry material which can be burned.

Concerns over incineration relate mainly to the by-products of the combustion process, most particularly the emissions to the atmosphere. Some pollutant emissions from incinerators are formed, in part, by incomplete combustion that may in turn lead to the formation of pollutants such as dioxins and furans. The formation of products of incomplete combustion is governed by the duration of the combustion process, the extent of gas mixing in the combustion chamber, and the temperature of combustion.

Outputs from incinerators include:

1. 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
2. Air pollution control residues (fly ashes)
3. Emissions of gaseous combustion products

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 environmental pressures related to incineration can be summarized as follows:

Air:	emission of particulate matter (PM), SO_2, NOx, HCL, HF, NMVOC, CO, CO2, N_2O, dioxins, furans, heavy metals (Zn, Pb, Cu, As, Ni, Hg, Cd)
Water:	deposition of hazardous substances on surface water
Soil:	landfilling of ashes
Ecosystem:	contamination and accumulation of toxic substances in the food chain

6. Landfill

Disposing of waste in a landfill is one of the most traditional methods of waste disposal, and it remains a common practice in most countries. In the past, landfills were often established in unused quarries, mining voids or borrow pits. Older and poorly-managed landfills can create a number of adverse environmental impacts such as wind-blown litter, rodents and other vermin, and the generation of leachate as a result of rain percolating through the waste and reacting with the products of decomposition, chemicals and other materials in the waste polluting groundwater and surface water. Another by-product of landfills is landfill gas (mostly composed of methane and carbon dioxide), which is produced as organic waste breaks down anaerobically. This gas can create odour problems, kill surface vegetation, and is a greenhouse gas.

The main potential impacts on health arise from landfill gas and exposure to groundwater contaminated by landfill leachate. Both gaseous and aqueous emissions from landfills are highly complex mixtures whose characteristics vary considerably from site to site and with waste composition and age of the landfill.

Emissions of landfill gas and leachate from biodegradable waste materials take place over a period of years following disposal. Landfill gas is the principal component of emissions to air from landfill sites. 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 and carbon dioxide. Small amounts of trace components such as organic gases or vapours are also present. There are a number of ways in which landfill gases and products of combustion are released into the atmosphere.

1. Fugitive gas emissions from passive venting into the atmosphere.
2. Collection using a gas extraction system and subsequent burning in flares.
3. Collection using a gas extraction system and utilised to provide heat or power using an energy recovery plant that uses the landfill gas as a flammable fuel.

The environmental pressures related to landfills can be summarized as follows:

Air: emissions of methane (CH4), carbon monoxide (CO), SO_2, HCL, HF, dioxins, furans,

Water: leaching of salts, heavy metals, biodegradable and persistent organics to groundwater

Soil: accumulation of hazardous substances in soil

Ecosystem: contamination and accumulation of toxic substances in the food chain

7. Transportation

Transport of waste, usually using heavy lorries, is a significant part of waste management practices, not only regarding the transport of raw waste to disposal sites, or transfer sites, but transport resulting from separation of the waste into more fractions for advanced treatment (higher distances for recycled materials). Waste transport accounts for 5% of the energy consumed by the transport sector and 15% of freight transport in tonne - kilometres.

Transportation of waste for both recycling and disposal uses lorries, especially with diesel engines, with the following environmental pressures (NSCA 2002):

Air: emission of particulate matter (PM), NOx.

APPENDIX 2

EU legislation on waste

The main European policy on waste has been defined in the Thematic Strategy on the prevention and recycling of waste proposed on 21 December 2005 (http://eur-lex.europa.eu/LexUriServ/site/en/com/2005/com2005_0666en01.pdf) as a follow up of the 6th Environment Action Programme (6th EAP), adopted by the European Parliament and Council in 2002.

The current waste policy aims to prevent waste and promote re-use, recycling and recovery so as to reduce the negative environmental impact. Current EU waste policy is based on a concept known as the “waste hierarchy”. This means that, ideally, waste should be prevented and what cannot be prevented should be re-used, recycled and recovered as much as feasible, with landfill being used as little as possible. Landfill is the worst option for the environment as it signifies a loss of resources and could turn into a future environmental liability. The aim of moving towards a recycling and recovery society means moving up the hierarchy, away from landfill and more and more to composting, recycling and recovery.

As already mentioned, under EU policy, landfilling is seen as the last resort and should only be used when all other options have been exhausted, i.e., only material that cannot be prevented, re-used, recycled or otherwise treated should be landfilled. It is to be noted that diversion of waste away from landfill is an important element in EU policy in order to improve the use of resources. In particular, with the aim of fulfilling the targets provided by Directive 1999/31/EC on Landfill of Waste, Member States are obliged to set up national strategies for reducing the amount of biodegradable municipal waste going to landfill. With these measures and with the general provision that only waste which has been subjected to treatment can be landfilled, the Landfill Directive is expected to have a major effect on the waste management system. This includes recovery of waste and possibly also prevention of waste.

On the other hand, although some progress has been made in the area of waste management in Europe (clean up of many old incinerators, implementation of new techniques, etc.) and waste prevention has been the objective of both national and EU waste management policies in recent years, limited progress has been made so far in transforming this objective into practical action.

In sum, the EU policy could be summarized as “less waste to landfill, more compost and energy recovery from waste, more and better recycling”. The European policy is expected to have implications for current practices in the Member States and to create new opportunities for waste management options other than landfill with a general move up the waste hierarchy.

A summary of the EU legislation on waste

http://www.wasteonline.org.uk/resources/InformationSheets/Legislation.htm

The European Union's waste legislation comprises three main elements:

- **horizontal legislation**, establishing the overall framework for the management of wastes, including definitions and principles
- **legislation on treatment operations**, such as landfill or incineration, which may set technical standards for the operation of waste facilities
- **legislation on specific waste streams**, such as waste oil or batteries, which may include for example measures to increase recycling or to reduce hazardousness

Directive	Publication year	Directive number
Horizontal		
1. Directive on Waste (Waste Framework Directive)	1975	75/442/EEC
2. Directive on Hazardous Waste	1991	91/689/EEC
3. Directive on waste	2006	2006/12/EC
Treatment		
4. Directive on Integrated Pollution Prevention and Control	1996	96/61/EC
5. Directive on the Landfill of Waste	1999	1999/31/EC
6. Directive on the Incineration of Waste	2000	2000/76/EC
Waste stream		
7. Directive on Batteries and Accumulators	1991	91/157/EEC
8. Directive on Packaging and Packaging Waste	1994	94/62/EC
9. Directive on End of Life Vehicles (ELV)	2000	2000/53/EC
1. Directive on Waste Electrical & Electronic Equipment	2002	2002/96/EC

APPENDIX 3

A short description of waste management in the three countries

UK:

The main legal framework for the waste strategy in England and Wales is set out in part V of the Environment Act (1995). The Landfill regulations (2002)implement the Landfill Directive 99/31. The Waste and Emissions Trading Act provides the basis for establishing Landfill Allowance Trading Scheme (LATS). A number of stakeholders are involved in developing waste related plans in England and Wales: Central Government and the Welsh Assembly, Regional Planning Bodies in England, Waste Planning Authorities at the local level and The Environment Agency.

The vision for dealing with English waste was described in Waste Strategy 2000 for England and Wales (in Wales replaced by 'Wise about Waste: The National Waste Strategy for Wales, 2002). It sets out targets for the reduction of waste sent to landfills. The waste strategy also includes targets for increasing waste recycling. There are further targets to reduce the amount of MSW landfilled. These arise from the landfill directive (England has agreed with the European Commission on a four-year derogation to meet the targets). Tradable allowances have been introduced to restrict the amount of biodegradable municipal waste sent to landfills. The government also accepted the recommendations of the 'Waste Not Want Not' report published in 2002; recommendations aim to reduce the growth rate in waste from 3% to 2% per annum; boost recycling by developing the infrastructure; increase choices for managing waste by creating economic incentives, as well as the incentive to reduce damage to the environment; stimulate innovation in waste treatment and waste management organisations.

The most important national policy instruments on waste in England and Wales involve Landfill Tax from 1996, Landfill Tax Credit Scheme and Landfill Allowance Trading Scheme for municipal waste launched in 2005 (LATS). The combined effect of these policies is intended to reduce the use of landfills and promote other recovery and recycling options.

In Scotland, The National Waste Strategy: Scotland (1999) sets the framework and policies for moving towards sustainable waste management. It was replaced by The National Waste Plan 2003, prepared by the Scottish Executive and the Scottish Environment Protection Agency (SEPA) in consultation with key stakeholders. It provides an integrated summary of the 11 Area Waste Plans that were identified as the Best Practicable Environmental Option (BPEO) for dealing with municipal solid waste. Its implementation is being funded by the Strategic Waste Fund.

The Northern Ireland Waste Management Strategy, "Towards Resource Management" (2006-2020), aims to move waste management away from landfills towards more sustainable practices. The Strategy contains specific non-statutory targets for the recycling and composting of municipal and non-municipal waste streams.

Italy:

The National Waste framework law in Italy was issued in 1997 (Legislative Decree 22/97; updated on April 29, 2006 by the legislative decree n. 152 "Environment Act"), transposing three of the main EU directives on waste: European Waste Framework Directive 75/442/EEC (modified by Directive 91/156/EEC); Directive on Hazardous Waste 91/689/EC, and Directive on Packaging and Packaging Waste 94/62/EC. Decree 22/97 implemented the integrated waste management policy set up by the European Waste Strategy; according to the decree the waste management system is based on preventing waste generation and material and energy recovery from waste. It also defined the responsibilities among the main actors of the national waste management system - regions that hold the responsibility for drawing up waste management plans to integrate waste collection, treatment and disposal within optimal management areas (ATO, Ambito Territoriale Ottimale); and local authorities (Autorità di Ambito) have the responsibility to organise municipal waste collection and management.

From January 1st, 2007 decree 152/06 sets targets about the weight of separate collections of municipal waste and by transposing the Directive 2004/12/EC, it improves MSW separate collection and recovery, redesigning the packaging waste management system, on the basis of the “polluter pays” principle and the “shared responsibility” among all involved.

A waste information system has been developed at the national level, based on the National Waste Inventory, established in 1994. Hazardous waste producers and managers are required to report yearly to the National Waste Inventory about managed waste quantities and categories. The Inventory has its headquarters at the Agenzia Nazionale per la Protezione dell'Ambiente e per i servizi tecnici (APAT) and regional seats at ARPAs (the Regional Environmental Protection Agencies). National Inventory of Waste is considered as an implementation tool of the Regulation 2150/2002/EC on waste statistics.

The landfill system in Italy was reorganized by the Legislative Decree 36/03 in 2003. It establishes the classification of the landfills (for hazardous waste, non-hazardous waste and for inert waste), and further specifies the type of waste going to landfills and costs involved in the operation of the sites. Additionally, according to art. 5 (1) of the Landfill Directive 1999/31/CE, Italy has developed a national strategy regarding the reduction of biodegradable waste going to landfills and all regions have to approve a proper program integrating the Regional Waste Management Plan.

Directive 2000/76/EC on waste incineration has been transposed into national legislation through Legislative Decree no. 133 of 11 May, 2005. This decree establishes provisions for waste incineration and co-incineration; provides measures and procedures to prevent or reduce, as much as possible, negative effects of waste incineration on the environment, in particular the pollution of air, soil, surface and groundwater, and the resulting risks to human health. It sets up operating conditions and emission limits for incineration plants and sampling and testing methods for pollutants deriving from them.

There are several different policy instruments applied in Italy for waste management, including both, market based instruments (e.g. Municipal Waste Tariff - 2000, Tax on Waste Disposal - 1996, Surcharges on purchase of certain goods - 1988), and administrative instruments (e.g. Ban on Landfilling - 1998, MSW Separate Collection - 1997 and Packaging collection system – 1998).

Slovakia:

Waste management in Slovakia is regulated by Act No. 223/2001 on wastes (amended by later regulations; currently Act No. 409/2006 Coll.) and by a set of implementing regulations. The act was put into effect on 1 March 2001 and has been harmonised with all EU Waste Directives, including the Directive 2000/53/EC on end-of life vehicles, the Directives on electrical and electronic equipment waste (WEEE), the Directive on PCB/PCT, the Directive on hazardous waste, the Landfill Directive 1999/31/EC and others. Furthermore, The Waste Act introduced market oriented economic instruments in the environmental legislation, including the establishment of the non-governmental Recycling Fund that provided more than 40 million EUR for the purposes of building up an infrastructure of waste management and facilities for recovery or recycling wastes.

The Waste Act No. 223/2001 Coll. provides the Ministry of Environment with a mandate to develop a National Waste Management Programme. Additionally, regional environmental offices and municipalities must operate their own waste management programmes, harmonised with the national programme. The Waste Management Plan of the SR for 2006-2010 (4^{th} plan in action), approved by the Government of SR on 15 February, 2006, is a basic planning document for waste management which covers entire waste management system of the country. Among other things, WMP contains information on total waste management and on management of waste streams (hazardous, municipal, biodegradable wastes, PCBs and packaging wastes), proposed measures to achieve objectives of the WMP for selected waste streams and definitions of recovery and recycling targets. The main strategy is to increase material and energy recovery of wastes and decrease landfilling to 13% according to total produced waste amount in 2010. On January 2004 came into force Act No. 17/2004 that presents an economical tool for decreasing of amounts of wastes deposited to landfills.

Directives on waste incineration (2000/76/EC, 89/369/EEC, 89/429/EEC, 94/67/EC) were transposed to the Act No 478/2002 on air protection and Order of the MoE 706/2002 on air pollution sources, emission limits, and general operational conditions on list of pollutants.

National policy instruments on waste represent Landfill Tax from 1992, Recycling Fund (2001) with mandatory contributions from producers of selected types of commodities, Packaging and Packaging Waste Act (Act No. 529/2002) transposing the EC Packaging and Packaging Waste Directive and Environmentally Motivated Subsidies from State Environmental Fund from 2004.

APPENDIX 4
Data Collection protocol

Municipal solid waste data by country

A. TOTAL REFERENCE POPULATION *(number of inhabitants)*

B. TOTAL TONS OF WASTE PRODUCED PER YEAR

C. WASTE COMPOSITION *(types of waste fractions)*

Example:

FERROUS

GLASS .

MISC NON-COMBUSTIBLES

NAPPIES AND SANITARY

NON-FERROUS METALS

PAPER

PLASTICS

TEXTILES

ORGANIC MATERIAL

Collection and transport data

D. TYPES OF COLLECTION

Example: collection of waste to be recycled, collection of waste to be incinerated, etc.

<u>Per each type of collection:</u>

E. TYPES OF VEHICLES USED

<u>Per each type of vehicles used:</u>

F. FUEL: DIESEL, GASOLINE, ELECTRIC

G. COLLECTION ROUTES

Example: 50% urban mode, 40% rural mode, 10% motorway mode

Treatment data by Country

H. AMOUNT OF WASTE BY TYPE OF TREATMENT

Example: recycling, landfill, incineration, etc.

The following information refers to single facilities in each country. For all the facilities, GIS coordinates should be provided.

<u>For landfills</u>

I. SPECIFICATIONS OF THE FACILITY:

a. TONNAGE STORED (TONS PER MONTH)

b. YEAR OPERATION BEGAN

c. YEAR OF CLOSURE

d. TREATMENT OF LANDFILL GAS (FLARE OR GENERATOR)

For incinerators

J. SPECIFICATIONS OF THE FACILITY:

a. TREATED TONNAGE (TONS PER MONTH)

b. YEAR OPERATION BEGAN

c. YEAR OF MAJOR CHANGES

d. ELECTRICITY CONSUMPTION (KWH PER MONTH)

e. MATERIAL OUTPUT

BOTTOM ASH (KG PER MONTH)

FLY ASH (TONNS PER MONTH)

f. ENERGY GENERATION

ELECTRICITY (KWH PER MONTH)

Information for air dispersion modelling

stack height (m),

stack diameter (m),

exit velocity (m/s),

emission rate (m3/s),

exit temperature (°C)

APPENDIX 5

LARGE SCALE MODELING THE IMPACT OF INCINERATORS

1 Assessment methodology

2.1.1. Large scale modelling

Within the case studies of human health impact due to emissions of MWI (municipal waste incinerators) many important but different issues have to be taken into account. A good reference to the full methodology is provided in (ExternE, Methodology 2005 Update). Methodology has be further developed with the NEEDS project (NEEDS 2008).
Within this section only the European wide dispersion and chemical transformation of so called "classical air pollutants" and their impact on human health is described and results are provided in Chapter regarding results. Calculations have been performed with EcoSenseWeb.

EcoSenseWeb is an integrated computer system developed for the assessment of environmental impacts and resulting external costs from electricity generation systems and other industrial activities. Based on the Impact Pathway Approach (IPA) developed in the ExternE-Project on External Costs of Energy funded by the European Commission, EcoSenseWeb provides relevant data and models required for an integrated impact assessment related to pollutants.

Modules for assessment of emissions to air, soil and water are also included. Comprising so called classical airborne pollutants, heavy metals, greenhouse gases and radio nuclides.

Different impact categories can be considered including human health, crops yield loss, damage to building materials, loss of biodiversity and climate change.

One of the major objectives of the EcoSenseWeb development was to produce a user friendly system that is capable of performing a highly standardised impact assessment procedure with a minimum of data required as input from the user. Only the technical data of the facility to be analysed has to be added by the user. All other data are provided by the system, thus the user loses no time by the tedious compilation of data. However, it is obvious that the approach of providing all important data and models to the user limits the flexibility of the system. Although the various modules of the system have a potential for high flexibility, the current EcoSenseWeb version is limited to a set of standard applications that can very easily be carried out. A basic decision during the design phase of the system with respect to an easy handling was the selection of a single co-ordinate system. The European wide grid used by the "Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air pollutants in Europe" (EMEP) with the spatial resolution of approximately 50 x 50 km2 (EMEP50 grid) was applied. The EcoSenseWeb and the calculation of physical impacts (like YOLL, i.e. years of life time lost) and external costs follow as far as possible the so called Impact Pathway Approach (IPA).

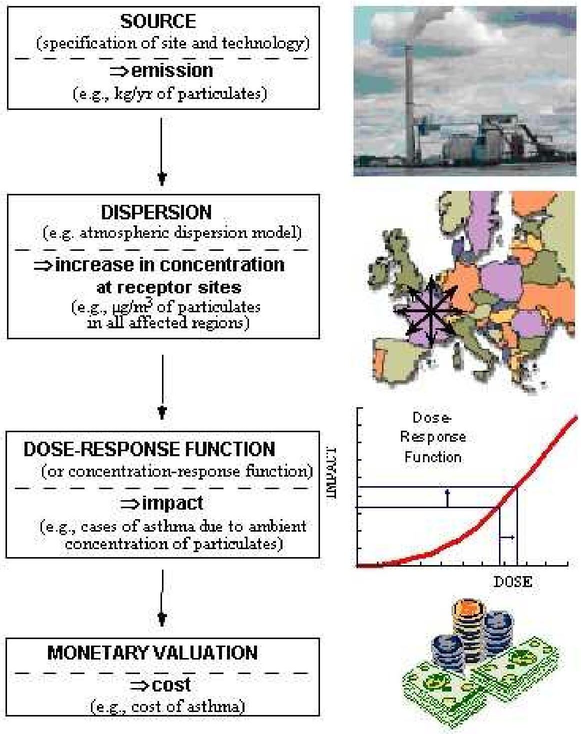
The link to the online tool EcoSenseWeb is http://EcoSenseWeb.ier.uni-stuttgart.de/ .

Introduction - the Impact Pathway Approach

For the calculation of site specific and marginal damages the Impact Pathway Approach should be applied for each source of emission, e.g. a MWI and the corresponding emissions from the stack.

Marginal damages have to be calculated because the creation of secondary pollutants like sulfates, nitrates and ozone depends also on the background concentration of NOx, SO2, NH3, NMVOC, etc. Therefore, two scenarios have to be calculated, one background and one with additional or reduced emissions.

Figure 1: Impact Pathway Approach



As shown in Figure 1, the IPA starts with the emission of a pollutant at the location of the source into the environment; models its dispersion and chemical transformation in the different environmental media; identifies the exposure of the receptors and calculates the related impacts which then are aggregated to external costs.

For many questions in research and policy we are not interested in the damages caused by one single process at a certain location but we are interested in the damages per functional unit of a certain technology or even per economic sector in different regions, countries or sub-regions. Secondly, the high quality dispersion modelling at regional, i.e. a European scale makes it very expensive, and if the data is not available, impossible, to perform calculations for each source of emission.

With regard to the classical pollutants, parameterised results from an Eulerian dispersion and chemical transformation model from The Norwegian Meteorological Institute (Tarrasón, 2008) have been derived based on source receptor matrices (SRM). These SRM allow attributing to each unit of emission in one region a concentration or deposition increment in each of the 50 x 50 km2 EMEP grid cells all over Europe. According to the IPA with the aid of CRF (concentration response functions) and the number of population physical impacts are then calculated for each grid cell. Finally, the impacts can be weighted and aggregated by means of monetary valuation of each physical impact in order to derive external costs per unit of emission.

The concentration increment for classical air pollutants in Europe correspond to different heights of release.

Results are available for emission of the following pollutants:

- NH3
- NMVOC
- NOX
- PM coarse
- PPM2.5
- SO2.

Results are available for emissions in 39 European countries and the 5 sea regions. The receptor domain covers the whole of Europe. Impacts included are impacts to human health, crops, damage to materials and loss of biodiversity. Population data is taken from (SEDAC 2006) - Gridded Population of the World.

Classical air pollutants - Dispersion & Chemical Transformation in Europe

Model runs were performed with the EMEP/MSC-West Eulerian dispersion and chemical transformation model. These model runs are based on emission scenarios, i.e. including spatial distribution of different sources on a 50 x 50 km2 EMEP grid. Model runs have been performed which take into account all source and all emission data. Secondly, model runs have been performed reflecting the emissions and corresponding impacts of only SNAP sector S1, i.e. "Combustion in energy and transformation industry", i.e. very high stacks. Moreover, meteorological conditions are included and since they differ from one year to another not only one but a set of representative meteorological years have been chosen by MET.NO. To derive the SRM a reduction of each pollutant by 15% for each source of emission within a corresponding sub-region is modelled separately. Europe is divided into 66 regions, i.e. some larger countries are subdivided into sub-regions.

For a 15% reduction of an airborne pollutant (e.g. NOx) within a country / sub-region of Europe (e.g. Belgium = BE) based on meteorological conditions (e.g. in the year 2000) and background emissions of e.g., the year 2010 a model run was performed by MET.NO. The result is a matrix covering the resulting concentration of different pollutants in each of the 50 x 50 km2 grid cell of the EMEP grid. This matrix contains the results in terms of concentrations of a primary (NOx) or secondary (nitrates and ozone, increased sulphates, etc.) air pollutants on the grid. The chemical reactions and interactions are quite complex. For example, a reduction of NOx emissions leaves in regions where NH3 is in the air, e.g. due to agricultural processes, more background NH3 for reaction with SO2 which was already in the background emitted, and therefore, increases the concentration of sulphates at some locations, etc.

Table 1 shows some of the primary and secondary pollutants and depositions provided by the regional and Northern Hemispheric dispersion model.

Shortcut	Comment	Unit
aNH_4	ammonium particles (ammonium nitrate and sulphate)	$\mu gN/m^3$
aNO_3	nitrate particles with diameter below 2.5µm	$\mu gN/m^3$
DDEP_OXN	total dry deposition of oxidized nitrogen	Mg/m^2
DDEP_RDN	total dry deposition of reduced nitrogen	Mg/m^2
DDEP_SOX	total dry deposition of sulphur	Mg/m^2

NO_x	$NO_x = NO_2 + NO$	µgN/m^3
pNO_3	nitrate particles with diameter above 2.5 a. below 10 µm	µgN/m^3
SIA	secondary inorganic aerosols	µg/m^3
SO_4	sulphate, includes also ammonium sulphate	µgS/m^3
SOMO35	sum Over Means Over 35 ppb	Ppb day
tNO_3	total coarse and fine nitrate aerosols	µgN/m^3
WDEP_OXN	wet deposition of oxidized nitrogen	Mg/m^2
WDEP_RDN	wet deposition of reduced nitrogen	Mg/m^2
WDEP_SO_x	wet deposition of sulphur	Mg/m^2
PPM_{25}	primary particles with diameter below 2.5 µm	µg/m^3
PPM_{co}	primary particles with diameter between 2.5 and 10 µm	µg/m^3

SIA includes all particles with an aerodynamic diameter < 10µm. It consists mainly of ammonium nitrate and sulphates. However, sulfates are mainly < than 2.5 µm. Therefore, SIA2.5 and SIAcoarse are derived to be able to apply the concentration response functions (CRF) regarding impacts to human health.

Meteorological years

Based on meteorological years 1996, 1997, 1998 and 2000 average results have been derived representing typical conditions of the present. This exercise has been performed in order to reflect not only one, more or less arbitrary year, but more typical and average conditions like wind speed, wind direction, precipitation, temperature, stability, etc.

The year 2003 was an exceptional warm year in Europe. Therefore, the results based on meteorological year 2003 can be used to estimate future conditions, reflecting the influence of climate change.

Background emission scenario 2010 and 2020

Since the background concentration of NH3, NMVOC, NOX and SO2 influences the creation of secondary pollutants (sulphates, nitrates, ozone) two set of SRM are available. One corresponds to conditions in 2010 and second corresponds to anticipated conditions in 2020. In general the emissions in 2020 are lower than in 2010. Because of non-linearity of the chemistry the creation of secondary pollutants and hence the marginal damage per unit of emission differs between the two scenarios.

It has to be emphasised that because of non-linear atmospheric chemistry and because of different background concentrations of e.g. NOx and NMVOC, especially with regard to ozone there can occur large differences in [Euro per tonne] values. Negative external costs can occur for NOx emission in 2010 but also for a view cells in 2020 values.

It is recommendable, especially with regard to cost benefit analysis for future measures to use the set corresponding to the conditions in 2020. If years between 2000 and 2014 are defined in the “Assumption” the dispersion results reflecting background emissions in 2010 are used. All other future years are based on 2020 background emissions.

For the results shown in this report the background emission of 2010 has been used because current operating plants have been assessed.

Height of release

The SRM have been derived by simulation of 15% emission reduction in each sub-region. This has been done in two ways, providing two sets of SRM, i.e.

- for pollutants from all sources, i.e. all SNAP sectors (i.e., including transport, industry, domestic firing systems, but also combustion plants), and
- for pollutants (primary particles, SO_2 and NO_x) from for SNAP sector 1 (combustion in power plants) only.

Further processing of the results allowed to derive values corresponding to low release height (< 100m) and high release heights (> 100m).

Physical Impacts to Human Health and external costs

According to the IPA the physical impact to the receptors can be calculated by multiplying the concentration or deposition in each grid cell with the number of receptors, e.g. population or surface area, and by a factor per unit of concentration or deposition. The impact over the whole area of Europe is than summed up. The methods are further described below.

Concentration Response Functions (CRF) and monetary values for Human Health impacts - Classical air pollutants

In Table 1 the overview of the concentration response functions for PM and ozone and corresponding monetary values are given. These are the most important and reliable concentration response functions (core). The corresponding monetary values demonstrate the different severity of the endpoints. Since the probability of the less severe endpoints is much higher they also contribute considerably to the total external costs. Nonetheless, the reduced life time expectancy (YOLL, Years of life lost) is the most important endpoint.

The set of so called core CRF is available from NEEDS project (Torfs et al, 2007). Since the secondary particles (nitrates and sulfates, Secondary Inorganic Aerosols) are assumed to be equally toxic as the Primary Particulate Matter this set of CRF is called “SIA_E_PPM”. However, the research and the assumptions of the toxicity of nitrates and sulphates, and the relation to NO_2 are still controversially discussed and sensitivity assessments are also possible.

For the purpose of INTARESE case study on waste incineration only the YOLL of classical air pollutants are assessed and reported in this section on large scale modelling.

Table 1: Overview of the concentration response functions for PM and ozone and corresponding monetary values.

Core Endpoints	Pollutant	Risk group (RG)	RGF value	Age Groupe (AG)	AGF value	CRF [1/(µg/m3)]	phys. Impact per person per µg per m3 [1/(µg/m3)]	unit	Monet Val per case or per YOLL [Euro]	External costs per person per µg per m3 [1/(µg/m3)]
primary and SIA < 2.5, i.e. Particle < 2.5µm										
Life expectancy reduction - YOLLchronic	PM2.5	all	1.000	Total	1	6.51E-04	6.51E-04	YOLL	40,000	2.60E+01
netto Restricted activity days (netRADs)	PM2.5	all	1.000	MIX	1	9.59E-03	9.59E-03	days	130	1.25E+00
Work loss days (WLD)	PM2.5	all	1.000	Adults_15_to_64_years	0.672	2.07E-02	1.39E-02	days	295	4.10E+00
Minor restricted activity days (MRAD)	PM2.5	all	1.000	Adults_18_to_64_years	0.64	5.77E-02	3.69E-02	days	38	1.40E+00
primary and SIA < 10, i.e. Particle < 10µm										
Increased mortality risk (infants)	PM10	infants	0.002	Total	0.009	4.00E-03	6.84E-08	cases	3,000,000	2.05E-01
New cases of chronic bronchitis	PM10	all	1.000	Adults_27andAbove	0.7	2.65E-05	1.86E-05	cases	200,000	3.71E+00
Respiratory hospital admissions	PM10	all	1.000	Total	1	7.03E-06	7.03E-06	cases	2,000	1.41E-02
Cardiac hospital admissions	PM10	all	1.000	Total	1	4.34E-06	4.34E-06	cases	2,000	8.68E-03
Medication use / bronchodilator use	PM10	Children meeting PEACE criteria - EU average	0.200	Children_5_to_14	0.112	1.80E-02	4.03E-04	cases	1	4.03E-04
Medication use / bronchodilator use	PM10	asthmatics	0.045	Adults_20andAbove	0.798	9.12E-02	3.27E-03	cases	1	3.27E-03
Lower respiratory symptoms (adult)	PM10	symptomatic_adults	0.300	Adults	0.83	1.30E-01	3.24E-02	days	38	1.23E+00
Lower respiratory symptoms (child)	PM10	all	1.000	Children_5_to_14_years	0.112	1.86E-01	2.08E-02	days	38	7.92E-01
Ozone [µg/m3] - from SOMO35 by multiplication by *1/365										
Increased mortality risk	SOMO35*1/365	Baseline_mortality	0.0099	Total (YOLL = 0.75a/case)	1	3.00E-04	2.23E-06	YOLL	60,000	1.34E-01
Respiratory hospital admissions	SOMO35*1/365	all	1.000	Elderly_65andAbove	0.158	1.25E-05	1.98E-06	cases	2,000	3.95E-03
MRAD	SOMO35*1/365	all	1.000	Adults_18_to_64_years	0.64	1.15E-02	7.36E-03	days	38	2.80E-01
Medication use / bronchodilator use	SOMO35*1/365	asthmatics	0.045	Adults_20andAbove	0.798	7.30E-02	2.62E-03	cases	1	2.62E-03
LRS excluding cough	SOMO35*1/365	all	1.000	Children_5_to_14_years	0.112	1.60E-02	1.79E-03	days	38	6.81E-02
Cough days	SOMO35*1/365	all	1.000	Children_5_to_14_years	0.112	9.30E-02	1.04E-02	days	38	3.96E-01

Whereas:

Risk Group RG: group within the general population with handicap

RGF value: Share of RG within the general population

Age group AG: groups distinguished by different age cohorts

AG value: Share of different age cohorts

CRF: concentration-response function

YOLL: years of life lost

RAD: Restricted activity days

WLD: Work loss days

MRAD: Minor restricted activity days

LRS: lower respiratory symptoms

Three CRF are considered to evaluate the number of YOLL.

Life expectancy reduction due to PM2.5

The increased mortality of infants due to PM10 and

The increased mortality due to ozone.

The CRF for PM2.5 calculates the YOLL directly. With regard to increased mortality of infants 80 YOLL per cases is assumed, with regard to increased mortality due to ozone 0.75 years per cases are assumed.

The CRF, e.g. for primary and secondary PM2.5 leads to the following results:

If 100,000 people are exposed to one additional µg of PM2.5 per m3 in the ambient air 65.1 life years will be lost in this group.

2. Results

2.1.2. Large scale modelling

As input data the emission data of classical pollutants, location and technical specifications of the waste incinerator plants in Italy, Slovakia and England has been used. In the following tables the results for the different MWI are listed. The Years of life time lost (YOLL) in Europe in each year of operation are calculated. These are caused by emission of

- NH3
- NMVOC
- NOX
- PM coarse
- PPM2.5
- SO2.

2.1.1. Italy

For Italy there were two sets of emission. One called "average values" and a second called "national limits". The national limits are larger than the "average values".

Some location has several lines which are indicated in the list by "line xy".

Location	Emissions	[YOLL] Years of life time lost
Arezzo_01	national limits	3.5243E-01
Arezzo_02	average values	2.0176E-01
Bologna_01	national limits	3.3278E+00
Bologna_02	average values	2.8233E+00
Bolzano_01	national limits	3.2077E+00
Bolzano_02	average values	2.1830E+00
Brescia_line1_01	national limits	3.7062E+00
Brescia_line1_02	average values	2.5411E+00
Brescia_line2_01	national limits	4.3596E+00
Brescia_line2_02	average values	2.9906E+00
Busto Arsizio_01	national limits	1.7260E+00
Busto Arsizio_02	average values	1.1744E+00
Ca'del Bue_01	national limits	1.7312E+00
Ca'del Bue_02	average values	1.1778E+00
Castelnuovo_01	national limits	3.8191E-01
Castelnuovo_02	average values	2.5983E-01

Como_01	national limits	2.2910E+00
Como_02	average values	1.5589E+00
Cremona_line1_01	national limits	9.4195E-01
Cremona_line1_02	average values	6.4082E-01
Cremona_line2_01	national limits	1.2220E+00
Cremona_line2_02	average values	8.3139E-01
Dalmine_01	national limits	1.4002E+00
Dalmine_02	average values	9.5266E-01
Desio_01	national limits	9.6743E-01
Desio_02	average values	6.5821E-01
Ferrara_Canalb_01	national limits	1.1456E+00
Ferrara_Canalb_01	average values	7.7947E-01
Ferrara_Conch_01	national limits	1.5275E+00
Ferrara_Canalb_01	average values	1.0392E+00
Forli_01	national limits	2.4278E-01
Forli_02	average values	1.3900E-01
Fusina_01	national limits	1.1456E+00
Fusina_02	average values	7.7947E-01
Livorno_01	national limits	1.1456E+00
Livorno_02	average values	7.7947E-01
Macchiareddu_01	national limits	3.5243E-01
Macchiareddu_02	average values	2.0176E-01
Macomer_line1_01	national limits	1.6446E-01
Macomer_line1_02	average values	9.4154E-02
Macomer_line2_01	national limits	1.6446E-01
Macomer_line2_02	average values	9.4154E-02
Melfi_line1_01	national limits	3.5243E-01
Melfi_line1_02	average values	2.0176E-01
Melfi_line2_01	national limits	4.3066E-01
Melfi_line2_02	average values	2.4665E-01
Mergozzo_01	national limits	4.5828E-01

Mergozzo_02	average values	3.1184E-01
Messina_01	national limits	3.5243E-01
Messina_02	average values	2.0176E-01
Milano_01	national limits	2.1399E+00
Milano_02	average values	1.4680E+00
Modena_line1_01	national limits	8.2740E-01
Modena_line1_02	average values	5.6296E-01
Modena_line2_01	national limits	8.2740E-01
Modena_line2_02	average values	5.6296E-01
Modena_line3_01	national limits	1.2729E+00
Modena_line3_02	average values	8.6601E-01
Montale Agliana_01	national limits	1.2729E+00
Montale Agliana_02	average values	8.6601E-01
Ospedaletto_01	national limits	1.1456E+00
Ospedaletto_02	average values	7.7947E-01
Padova_01	national limits	1.1456E+00
Padova_02	average values	7.7947E-01
Parona_01	national limits	1.1456E+00
Parona_02	average values	7.7947E-01
Piacenza_01	national limits	1.0693E+00
Piacenza_02	average values	7.2754E-01
Poggibonsi_01	national limits	2.2258E-01
Poggibonsi_02	average values	1.2746E-01
Ravenna_01	national limits	5.5762E-01
Ravenna_02	average values	2.4665E-01
Reggio_01	national limits	1.2831E+00
Reggio_02	average values	8.7301E-01
Rimini_line1+2_01	national limits	2.7412E-01
Rimini_line1+2_02	average values	1.5693E-01
Rimini_line3_01	national limits	4.6987E-01
Rimini_line3_02	average values	2.6903E-01

Rufina_01	national limits	3.5243E-01
Rufina_02	average values	2.0176E-01
Scarlino_01	national limits	3.5243E-01
Scarlino_02	average values	2.0176E-01
Schio_line1_01	national limits	3.0554E-01
Schio_line1_02	average values	2.0780E-01
Schio_line2_01	national limits	4.8367E-01
Schio_line2_02	average values	3.2906E-01
Schio_line3_01	national limits	4.8367E-01
Schio_line3_02	average values	6.2350E-01
Sesto S. Giovanni_01	national limits	5.6013E-01
Sesto S. Giovanni_02	average values	3.8109E-01
Statte_01	national limits	1.9578E-01
Statte_02	average values	1.1209E-01
Terni_01	national limits	1.5663E-01
Terni_02	average values	8.9683E-02
Tolentino_01	national limits	3.5243E-01
Tolentino_02	average values	2.0176E-01
Trieste_line1_01	national limits	1.3391E+00
Trieste_line1_02	average values	9.1105E-01
Trieste_line2_01	national limits	1.2475E+00
Trieste_line2_02	average values	8.4872E-01
Valmadrera_01	national limits	1.7820E+00
Valmadrera_02	average values	1.2125E+00
Vercelli_01	national limits	2.0362E+00
Vercelli_02	average values	1.3857E+00

2.1.2. Slovakia

In case of Slovakia only 2 different locations could be investigated. Data sets for before and after 2002 are available. For Bratislava 2 lines, or furnaces were in use.

Location	Emissions	[YOLL] Years of life time lost
Bratislava_after2002_01	furnace 1	5.7518E-01

Bratislava_after2002_02	furnace 2	9.4821E-01
Bratislava_before2002_01	furnace 1	2.7439E+00
Bratislava_before2002_02	furnace 2	5.4869E+00
Kosice_after_ 2002	--	9.2474E-01
Kosice_before_ 2002	--	1.1230E+00

2.1.3. UK

In case of England the location is give by the abbreviation, e.g. cryptic like AP3435SD. However, this corresponds to a certain MWI plat in England.

Location	[YOLL] Years of life time lost
AP3435SD	6.9817E-01
BJ6178IX	6.1708E-01
BJ7093IY	1.0996E+00
BJ7107IJ	1.2163E+00
BJ7786IV	5.0711E-01
BM4082IY	7.9579E-01
BR4551IC	3.9810E+00
BS3042IM	6.2471E-01
BT4249IB	3.0171E-01
EP3034SN	8.1439E-01
NP3738SY	2.1841E+00
NP3739PD	1.1980E+00
QP3234SX	6.9076E-01
VP3034SG	1.0706E+00
WP3239SJ	1.4428E+00
YP3033BE	3.2715E+00
YP3634SJ	3.6321E+00

3. *Uncertainty*

The following summary is taken from the deliverable (Spadaro and Rabl, 2007). In the deliverable “Report on the methodology for the consideration of uncertainties“ (Spadaro and Rabl, 2007) the issue of uncertainty is discussed and guidance on how to deal with uncertainty is provided.

Whereas the uncertainty of environmental impacts and damage costs is usually estimated by means of a Monte Carlo calculation, this paper shows that most (and in many cases all) of the uncertainty calculation involves products and/or sums of products and can be accomplished with an analytic solution which is simple and transparent. We present our own assessment of the component uncertainties and calculate the total uncertainty for the impacts and damage costs of the classical air pollutants. The distribution of the damage costs is approximately lognormal and can be characterized in terms of geometric mean μg and geometric standard deviation σg, implying that the confidence interval is multiplicative. We find that for the classical air pollutants σg is approximately 3 and the 68% confidence interval is [μg / σg, μg x σg]. In other words, with 68% probability the value is in the range of 3 times or one third of the reported values. Because the lognormal distribution is highly skewed for large σg, the median is significantly smaller than the mean. We also consider the case where several lognormally distributed damage costs are added, for example to obtain the total damage cost due to all the air pollutants emitted by a power plant, and we find that the relative error of the sum can be significantly smaller than the relative errors of the summands. Even though the distribution for such sums is not exactly lognormal, we present a simple lognormal approximation that is quite adequate for most applications.

References

EcoSenseWeb, User's Manual

Preiss, Ph. and Klotz V., 2008, http://ecosenseweb.ier.uni-stuttgart.de/

ExternE, Methodology 2005 Update

European Commission (2005). ExternE – Externalities of Energy – Methodology 2005 Update. EUR 21951 EN. Office for Official Publications of the European Communities, Luxembourg. ISBN 92-79-00423-9.

NEEDS 2008

European Commission (EC) – Integrated project NEEDS – New Energy Externalities Developments for Sustainability, Project no: 502687, Research Stream 1b: "Development and improvement of a methodology to estimate external costs of energy" - http://www.needs-project.org/ .

SEDAC, 2006

Gridded Population of the World, version 3 (GPWv3), Center for International Earth Science Information Network (CIESIN), Columbia University; and Centro Internacional de Agricultura Tropical (CIAT). 2005. Gridded Population of the World Version 3 (GPWv3). Palisades, NY: Socioeconomic Data and Applications Center (SEDAC), Columbia University. Available at http://sedac.ciesin.columbia.edu/gpw , Feb 2007.

Spadaro and Rabl, 2007

NEEDS project, FP6, Rs1b_D7.2 "Report on the methodology for the consideration of uncertainties", ARMINES/Ecole des Mines, 60 boul. St.-Michel, 75272 Paris 06, Joseph V. Spadaro, spadarojv@aol.com and Ari Rabl, ari.rabl@ensmp.fr, 2007.

Tarrasón, 2008

Tarrasón, L., NEEDS project, FP6, Rs1b_TP1.2, ""Documentation and evaluation of the source-receptor calculations made for NEEDS"", Norwegian Meteorological Institute – Research and Development Department – Section Air Pollution, in progress., 2008.

Torfs et al, 2007

Torfs R., Hurley F. and Miller B. (2007), NEEDS project, FP6, Rs1b_D3.7, "Final Report on casual links between pollutants and health impacts – A set of concentration-response functions".

Appendix 6

Systematic Review of Epidemiological Studies on Health Effects Associated with Waste Management

Introduction

The “waste management”, that is the generation, collection, processing, transport, and disposal of municipal solid waste (MSW) is important for both environmental reasons and public health. There are a number of different options available for the management and treatment of waste including minimisation, recycling, composting, energy recovery and disposal. At present, an increasing amount of the resources contained in waste is recovered as materials but a large part is incinerated or permanently lost in landfills. The various methods of waste management release a number of substances, most in small quantities and at extremely low levels. However, concerns remain about potential health effects associated with the main waste management technologies. Because of the wide range of pollutants that may be released by the various management options, the different pathways of exposure, usually long-term low-level character of exposure, and a potential for synergistic and cumulative effects, there are many uncertainties involved in the assessment of health effects.

Several studies on the possible health effects for populations living in proximity of landfills and incinerators have been published and well-conducted reviews are available (Vrijheid, 2000, Rushton, 2003, Franchini et al, 2004; WHO, 2007). Associations with some reproductive and cancer outcomes have been suggested for both landfills and incineration. However, the reviews indicate the weakness of the results of these studies due to design issues, mainly related to lack exposure information, use of surrogate indirect measures such distance from the source, lack of control for potential confounders. As results, the controversy over the possible health effects of waste management in the public is large for many reasons that include risk communication, risk perception and conflicting interests of various stakeholdres. Therefore, there is a need of an appropriate risk assessment that informs both policy makers and the public with the currently available level of information on the health risks associated with the different waste management technologies. Of course, the current level of uncertainties should be taken into account.

Within the EU funded INTARESE project (Briggs, 2008), we aimed to assess potential exposures and health effects arising from municipal solid wastes, from generation to disposal or treatment. A key part in the health impact assessment was the selection or the development of a suitable set of relative risks that link individual exposures with specific health endpoints. We have then conducted a systematic review of epidemiologic literature on health effects associated with collecting, recycling, composting, incinerating, and landfilling of municipal solid waste with the specific aim to derive appropriate relative risk estimates associated to various waste management technologies. The levels of scientific uncertainties associated to these estimates have been estimated.

Methods

We have considered epidemiological studies conducted on the general population with potential exposures from collecting, recycling, composting incinerating, and landfilling municipal solid waste. We considered also studies of workers employed in waste management plants as they may be exposed to the same potential hazards as the community residents, even if the intensity and duration of the exposure may differ. Studies on populations with potential exposures from toxic and hazardous wastes were excluded as the exposures are different and not completely comparable to those arising from municipal wastes (Russi et al, 2008) Furthermore, we did not consider studies on biomarkers of exposure and health effects.

Relevant papers were found through computerized literature searches on the MEDLINE and PubMed Databases from 1/1/1983 through 31/12/2006, using the MeSH terms "waste management" and "waste products" and "health effects". We obtained 427 papers with this method. We also conducted a free search with several combinations of relevant key words ("waste incinerator or landfill or composting or recycling" and "cancer or respiratory effects or birth outcome or health effects"), and 224 papers were obtained. In addition, articles were traced through references listed in previous reviews (Poulsen et al, 1995a-b, Vrijheid 2000, Hu et al, 2001, Dolk et al, 2003, Rushton 2003, Franchini 2004) and in publications of the UK Department for Environment, Food and Rural Affairs (Enviros-DEFRA 2004).

All papers were independently evaluated by three observers about eligibility, and disagreements were resolved by discussion. As indicated, studies on industrial, toxic or hazardous waste, on sewage treatment or on biological monitoring were not included. We also excluded articles in languages other than English, not journal articles, and six studies (Rydhstroem et al, 1998, Fukuda et al, 2003, Altavista et al, 2004, Biggeri et al, 2005, Minichilli et al, 2005, Bianchi et al, 2006) conducted at municipality level (usually small towns) where it was not possible to evaluate the extent of the population potentially involved and the possibility of exposure misclassification was high.

Papers have been grouped according to the following criteria:

- waste management technologies: recycling, composting, incinerating, landfill (the term landfill is used here only for controlled disposal of waste land);
- health outcomes: cancers (stomach, colorectal, liver, larynx and lung cancer, soft tissue sarcoma, kidney and bladder cancer, non Hodgkin's lymphoma, childhood cancer), birth outcomes (congenital malformations, low birth weight, multiple births, abnormal sex ratio of newborns), respiratory, skin and gastrointestinal symptoms or diseases.

For each paper, we have reported in appropriate tables (in the appendixes) study design (e.g. geographical, cohort, cross-sectional, case-control study, etc.), population characteristics (subjects, country, age, sex), exposure measures (e.g. occupational exposure to municipal waste incinerator by-products, residence near a MSW landfill, etc.), and the main results (incl. control for major confounders) with respect to the quantification of the health effects studied. For each study we have evaluated the potential sources of uncertainty in the results due to design issues. In particular, the possibility that selection bias, information bias, or confounding could artificially increase or decrease the relative risk estimate has been noted in the tables using the plus/minus scale to indicate that effect estimates are likely to be overestimated (or underestimated) up to 20% (+/-), from 20 to 50% (++/--) and more than 50% (+++/---). Scoring of the uncertainties was done by two observers (SM and FF) with a discussion over the inconsistencies.

After the description of the available studies, the overall evaluation of the epidemiological evidence regarding the process/disease association was done on the basis of the IARC (1999) criteria, and two categories were chosen, namely: "*Inadequate*" when the available studies are of insufficient quality, consistency, or statistical power to decide the presence or absence of a causal association, or no data on cancer in humans are available; "*Limited*" when a positive association has been observed between exposure and disease for which a causal interpretation is considered to be credible, but chance, bias, or confounding could not be ruled out with reasonable confidence. In order to derive appropriate relative risks, only when the specific association was judged as limited, we considered the set of studies providing the best evidence and assigned an overall level of scientific confidence of the specific effect estimate based on an arbitrary scale: very high, high, moderate, low, very low. This process was done by three assessors (SM, DP, and FF) with a discussion over the inconsistencies.

Results

A total of 42 papers were reviewed: 28 concerning health effects in communities living in proximity to waste sites, 14 on employees working at waste management sites. The majority of community studies evaluated possible adverse health effects in relation to incinerators and landfills. We did find little evidence on potential health problems resulting from environmental or occupational exposures from composting or recycling, and very little on storage/collection of solid waste. A description of the main findings follows. The appendixes contain several details of the studies reviewed.

Studies of communities living near landfills

Nine epidemiological studies conducted on residents of communities near a landfill for municipal solid waste are reviewed and their main characteristics are listed in appendix 1.

Cancer

Four studies analysed the cancer risk of populations living near landfills. Michelozzi et al (1998) investigated the mortality risk in a small area of Italy (Malagrotta, Rome) with multiple sources of air contamination (a very large waste disposal site serving the entire city of Rome, a waste incinerator plant, and an oil refinery plant). Standardised Mortality Ratios (SMRs) were computed in bands of increasing distance from the plants, up to a radius of 10 km. No association between proximity to the sites and cancer of various organs, in particular liver, lung, and lymph haematopoietic cancer, was found; however, mortality from laryngeal cancer declined with distance from the pollution sources, and a statistically significant trend remained after adjusting for a four level index of socio-economic status. The main uncertainty of the study is related to the exposure assessment (--) since only distance was considered decreasing the possibility to detect an effect. There are also uncertainties in using mortality to estimate cancer incidence in proximity to a suspected source (+). On the other hand, even though the authors did adjust for an area-based index of deprivation, a residual confounding (+) from socioeconomic status is likely.

In a case-control study, Goldberg (1999) investigated cancer incidence among persons who had been living near a municipal solid waste landfill site in Montreal (Quebec, Canada). The exact street address at the time of diagnosis was used to classify subjects by geographic zones and distance from the site. The results of the analyses suggest a possible association for liver, kidney, pancreatic cancer, and non-Hodgkin's lymphomas with a borderline statistical significance. In this study, participation rate was greater for cases than for controls with a slight possibility of a selection bias (+) whereas there are some possibilities that misclassification in the exposure assessment would have diluted the effect estimates (-).

In Finland, Pukkala et al. (2001) studied whether the exposure to landfill caused cancer or other chronic diseases in inhabitants of houses built on a former dump area containing industrial and household wastes. After adjustment for age and sex, an excess number of male cancer cases was seen, especially for cancers of the pancreas and of the skin. The relative risk slightly increased with the number of years lived in the area. Some uncertainties are likely to affect the results of the study with regards to the exposure assessment (-), outcome assessment (+) and presence of residual confounding (-).

Jarup et al. (2002) examined cancer risks in populations living within 2 km of 9,565 (from a total of 19,196) landfill sites that were operational at some time from 1982 to 1997 in Great Britain). No excess risks of cancers of the bladder and brain, hepatobiliary cancer or leukaemia were found, after adjustment for age, sex, calendar year and deprivation. The study is very large and with a high power, however misclassification of exposure could have decreased the possibility to detect an effect (--).

In summary, there is inadequate evidence of an increased risk of cancer for communities living in proximity of a landfill. The two slightly positive studies from Goldberg et al (1999) and Pukkala et al (2001) are not consistent, although pancreatic cancer was elevated in both investigations.

Birth defects and reproductive disorders

Five studies examined reproductive effects of landfill emissions. We did not include in the present review the EUROHAZCON study (Dolk et al, 1998), the most comprehensive multi-site study of congenital anomalies in the vicinity of hazardous waste landfill sites in Europe, as we were dealing with municipal solid waste disposal.

The most important research in this field is that by Elliott et al. (2001). This study investigated the risk of adverse birth outcomes in populations living within 2 km of 9,565 landfill sites in Great Britain, operational at some time between 1982 and 1997, compared with those living further away (reference population). The sites comprised 774 sites for special (hazardous) waste, 7803 for non-special waste and 988 handling unknown waste, and the 2 km zone was constructed around each site to give resolution at the likely limit of dispersion for landfill emissions, including 55% of the national population. Among 8.2 million live births and 43,471 stillbirths, 124,597 congenital anomalies (including terminations) were examined: neural tube defects, cardiovascular defects, abdominal wall defects, hypospadias and epispadias, surgical correction of gastroschisis and exomphalos; low and very low birth weights were also analysed, defined as less than 2500 g and less than 1500 g, respectively. The main analysis was for all landfill sites for the combined period during their operation and after closure, and additional analysis was carried out separately for sites handling special waste and non-special waste, and in the period before and after opening for the 5,260 landfills with available data. We have reported the results for non-special waste sites in the appendix. After adjustment for deprivation and other potential confounding variables (sex, year of birth, administrative region), there was a small increase in the relative risks for low and very low birth weight and for all congenital anomalies, except for cardiovascular defects. There was no excess risk of stillbirth.

In a study published in 2000, Fielder et al. (2000) found that residents living near the Welsh landfill of Nant-y-Gwiddon in Wales had an increased risk of having a baby with a congenital malformation, not only after the site became operational but also before. Palmer et al. (2005), however, examined rates of congenital anomalies in births to mothers living within 2 km of 24 landfill sites in Wales, and found a significant increase in birth defects after the sites were opened.

The studies conducted in England suffer from the same limitations, namely the possibility that misclassification of exposure could have decreased the relative risk estimates to some extent (--); on the other hand, there are several uncertainties related to the quality of registration of congenital malformations. In the latter case, a positive bias is more likely (++).

In Denmark, Kloppenborg et al. (2005) marked the geographical location of 48 landfills (Geographical Information System) and used maternal residence as the exposure indicator in a study of congenital malformations. The authors found no association between landfill location and congenital anomalies combined or of the nervous system, and a small excess risk for congenital anomalies of the cardiovascular system. Potential confounding from socioeconomic status is the major limitation of this study (+++).

In a retrospective cohort study, Gilbreath et al. (2006) evaluated adverse birth outcomes in infants whose birth records indicated maternal residence in 197 Alaska Native villages containing open dumpsites (solid waste sites that are not maintained, contain uncovered wastes, and have no boundaries). Outcomes included low and very low birth weight, preterm birth, and intrauterine growth retardation: The dump sites were categorized into high, intermediate, and low hazard categories. Results indicated a higher proportion of low birth weight infants from mothers in villages with intermediate and high hazard dumpsites, and more infants born to mothers from these villages suffered from intrauterine growth retardation. Although lacks more detailed exposure information (-), the study is of a very high quality and several potential confounders were considered.

In summary, an increased risk of congenital malformations and of low birth weight has been reported from studies conducted in England. The main uncertainty from these studies is completeness of the registration of birth defects. There is supportive evidence for an increased risk of low birth weight from the study conducted in Alaska (Gilbreath et al, 2006).

<u>Respiratory diseases</u>

A study conducted by Pukkala et al. (2001) in Finland evaluated prevalence of asthma in relation to the residence in houses built on a former dump area containing industrial and household wastes. Prevalence of asthma was significantly higher in the dump cohort then in reference cohort (living nearby but clearly outside the landfill site), and the increased relative risk of asthma did not vary by time after residents moved into dump site houses, nor with years lived in those houses. Unfortunately, there are no replications of this observation and the overall evidence may be considered inadequate.

Studies of landfills workers

Only one study on landfill workers was reviewed. Gelberg et al. (1997) conducted a cross-sectional study to examine acute health effects among employees working at the New York City Department of Sanitation, focusing on Fresh Kills landfill employees. Telephone interviews conducted with 238 landfill and 262 off-site male employees asked about potential exposures both at home and work, health symptoms for the previous six months, and other information (social and recreational habits, socio-economic status). Landfill workers reported a significantly higher prevalence of work-related respiratory, dermatological, neurologic and hearing symptoms than controls; the respiratory symptoms, as well as the dermatologic ones, were not associated with any specific occupational title or work task, other than working at the landfill, and the association remained, even when controlling for smoking status.

Studies of communities living near incinerators

Eighteen epidemiologic studies conducted on residents of communities with municipal solid waste incinerators have been reviewed and their characteristics are listed in Appendix 2.

Cancer

Eight studies analysed the cancer risk of emissions from incinerators, usually of old generation with high polluting characteristics.

Elliott et al. (1996) investigated cancer incidence between 1974 and 1987 among over 14 million people living near 72 solid waste incinerator plants in Great Britain. Data on cancer incidence among the residents, obtained from the national cancer registration scheme, were compared with national cancer rates, and numbers of observed and expected cases were calculated after stratification by a deprivation index, based on 1981 census. Observed-expected ratios were tested for decline in risk with distance up to 7.5 km. The study was conducted in two stages: the first involved a stratified random sample of 20 incinerators and, based on the findings, a number of cancers were then further studied around the remaining 52 incinerators (second stage). Over the two stages of the study there was a statistically significant ($p<0.05$) decline in risk with distance from incinerators for all cancers, stomach, colorectal, liver and lung cancer. The use of distance as the exposure variable in this study could have led to some degree of misclassification (--). On the other hand, the same authors observed that residual confounding (+) as well as misdiagnosis (+) might have contributed to increase the risk estimates. When further analysis were undertaken, including histological review of liver cancer cases (Elliott et al., 2000), the risk estimates were somehow reduced (0.53- 0.78 excess cases per 10^5 per year within 1 km, instead of 0.95 excess cases per 10^5 as previously estimated).

Biggeri et al. (1996) conducted a case-control study in Trieste to investigate the relationship between multiple sources of environmental pollution and lung cancer. Based on distance from the sources, spatial models were used to evaluate the risk gradients and the directional effects separately for each source, after adjustment for age, smoking habits, likelihood of exposure to occupational carcinogens, and levels of air particulate. The results showed that the risk of lung cancer was inversely related to the distance from the incinerator, with a high excess relative risk very near the source and a very steep decrease moving away from it. The main problem of the study is the difficulty to separate the effects of other sources of pollution based on distance and the possibility of a potential confounding from other sources remains (++).

Using data on municipal solid waste incinerators from the initial study by Elliott et al. (1996), Knox (2000) examined a possible association between childhood cancers and industrial emissions, including those from incinerators. From a file of 22,458 cancer deaths occurring in children before their 16th birthday between 1953 and 1980, he extracted 9,224 cases known to have moved at least 0.1 km between birth and death, and using a newly developed technique of analysis, distances from the suspected sources to the birth addresses and to the death addresses, respectively, were compared. The childhood-cancer/leukaemia data showed highly significant excesses of migrations away from birthplaces close to municipal incinerators, but the specific effects of the municipal incinerators could not be separated clearly from those of nearby industrial sources of combustion. Misclassification of exposure is the main limit of this paper (--).

In France, Viel et al. (2000) detected a cluster of patients with non-Hodgkin’s lymphoma and soft tissue sarcoma around a French municipal solid waste incinerator with high dioxin emissions (16.3 ng international toxic equivalency factor/m^3). To better explore the environmental origin of the cluster suggested by these findings, Floret et al. (2003) carried out a population-based case-control study in the same area, comparing 222 incident cases of non-Hodgkin’s lymphoma diagnosed between 1980 and 1995 and controls randomly selected from the 1990 population census. The risk of developing lymphomas was 2.3 times higher among individuals living in the area with the highest dioxin concentration than among those in the area with the lowest concentration. Given that a model was used to attribute exposure to cases and controls, a random misclassification could have reduced the effect estimates (--).

In Italy, a case-control study by Comba et al. (2003) showed a significant increase in risk of soft tissue sarcomas associated with residence within 2 km of an industrial waste incinerator in the city of Mantua, with a rapid decrease of risk at greater distances. There is a slight likelihood that an increased attention to the diagnosis for this form of cancer in the vicinity of the plant could have introduced a small bias (+) in the risk estimate.

Finally, an excess risk of lung cancer was also found in females living in two areas of the Province of La Spezia (Italy) exposed to environmental pollution emitted by multiple sources, including a waste incinerator (Parodi et al., 2004). Again in this study, a limited exposure assessment could have decreased the risk estimates (--) but positive confounding from other sources is very likely.

In summary, although several uncertainties limit the overall interpretation of the findings, there is limited evidence that people living in proximity of an incinerator have increased risk of all cancer, stomach, colon, liver, lung cancers based on the studies of Elliott et al (1996). Specific studies on incinerators in France and in Italy suggest an increased risk for non-Hodgkin lymphoma, and soft-tissue sarcoma.

Birth defects and reproductive disorders

Six studies examined reproductive effects of incinerator emissions (see Appendix 2).

Jansson et al. (1989) analysed whether the incidence of cleft lip and palate in Sweden has increased since refuse incineration started: the results of a register study, based on information from the central register of malformations and the medical birth register, did not demonstrate an increased risk.

A study by Lloyd et al. (1988) examined the incidence of twin births between 1975 and 1983 in two areas near a chemical and a municipal waste incinerator in Scotland: after adjustment for maternal age, an increased frequency of twinning in areas exposed to air pollution from incinerators was seen. In the same study areas, Williams et al. (1992) investigated sex ratios, at various levels of geographical detail and using 3-dimensional mapping techniques: analyses in the residential areas at risk from airborne pollution from incinerators showed locations with statistically significant excesses of female births.

To investigate the risk of stillbirth, neonatal death, and lethal congenital anomaly among babies of mothers living close to incinerators (and crematoriums), Dummer et al. (2003) conducted a geographical study in Cumbria (Great Britain). After adjustment for social class, year of birth, birth order, and multiple births, there was an increased risk of lethal congenital anomaly, in particular spina bifida and heart defects.

Subsequently, Cordier et al. (2004) studied communities with fewer than 50,000 inhabitants surrounding the 70 incinerators that operated at least one year from 1988 to 1997. Each exposed community was assigned an exposure index based on a Gaussian plume model, estimating inhalation per number of year the plant had operated. The results were adjusted for year of birth, maternal age, department of birth, population density, average family income, and when available, local road traffic. The rate of congenital anomalies was not significantly higher in exposed compared with unexposed communities; only some subgroups of congenital anomalies, specifically facial cleft and renal dysplasia, were more frequent in the exposed communities.

Tango et al. (2004) investigated the association of adverse reproductive outcomes with mothers living within 10 km from 63 municipal solid waste incinerators with high dioxin emission levels (above 80 ng international toxic equivalents TEQ/m^3) in Japan. To calculate the expected number of cases, national rates based on all live births, foetal deaths and infant deaths occurred in the study area during 1997-1998 were used and stratified by potential confounding factors available from the corresponding vital statistics records: maternal age, gestational age, birth weight, total previous deliveries, past experience of foetal deaths, and type of paternal occupation. None of the reproductive outcomes studied showed statistically significant excess within 2 km from the incinerators, but a statistically significant peak-decline in risk with distance from the incinerators was found for infant deaths and for infant deaths with congenital anomalies, probably due to dioxin emissions from the plants.

In sum, there are multiple reports of increased risk of congenital malformations among people living close to incinerators but there are no consistencies over the investigated outcomes. The overall evidence may be considered as limited. The study by Cordier at al

(2004) provides the basis for risk quantifications at least for facial cleft and renal dysplasia. Quantification for other reproductive disorders is more difficult.

Respiratory and skin diseases or symptoms

Four studies examined respiratory and/or dermatologic effects of incinerator emissions (see Appendix 2).

Hsiue et al. (1991) evaluated the effect of long-term air pollution resulting from wire reclamation incineration on respiratory health in children. 382 primary school children who resided in one control and three polluted areas in Taiwan were chosen for this study, and the results revealed a decrement in pulmonary function (including forced vital capacity and forced expiratory volume in 1 s) of those residents in the vicinity of incineration sites.

Shy et al. (1995) studied the residents of three communities having, respectively, a biomedical and a municipal incinerator, and a liquid hazardous waste-burning industrial furnace, and then compared results with three matched-comparison communities. After adjustment for several confounding (age, sex, race, education, respiratory disease risk factors), no consistent differences in the prevalence of chronic or acute respiratory symptoms resulted between incinerator and comparison communities. Additionally, no changes in pulmonary function between subjects of an incinerator community and those of its comparison community resulted from the study by Lee et al. (1999), based on a longitudinal component from the Health and Clean Air study by Shy et al. (1995).

Miyake et al. (2005) examined the relationship between the prevalence of allergic disorders and general symptoms in Japanese children and the distance of schools from incineration plants, measured using geographical information systems. After adjustment for grade, socio-economic status and access to health care per municipality, decreases in the distance of schools from the nearest municipal waste incineration plant was associated with an increased prevalence of wheeze and headache; there was no evident relationship between the distance of schools from such a plant and the prevalence of atopic dermatitis. The main factors that may affect the relative risk estimates in this study may be considered both reporting bias (++) and residual confounding from socioeconomic status (++).

In sum, although the intensive study conducted by Shy et al (1995) did not show respiratory effect, there are some indications of an increased risk of respiratory diseases, especially in children. However, the uncertainty related to outcome assessment and residual confounding is very high and the overall evidence may be considered as inadequate.

Occupational studies on incineration workers

Four studies conducted on incinerator workers were reviewed (see Appendix 3).

In 1997, Rapiti et al. conducted a retrospective mortality study on 532 male workers employed at two municipal waste incinerators in Rome (Italy) between 1962 and 1992. Standardized mortality ratios (SMRs) and their 90% confidence intervals were computed using regional population mortality rates. Mortality from all causes resulted significantly lower than expected (SMR=0.71; 90% CI=0.51-0.95), and all cancer mortality was comparable with that of the general population (SMR=0.95; 90% CI=0.58-1.46). Mortality from lung cancer was reduced, but increased risk was found for gastric cancer: analysis by latency indicated that this excess risk was confined in the category with more than 10 years since first exposure.

Bresnitz et al. (1992) studied 89 of 105 Philadelphia incinerator male workers, employed at the time of the study in late June 1988. Based on a work site analysis, workers were divided into potential high and low exposure groups, and no statistically significant differences in pulmonary function were found between the two groups, after adjustment for smoking status.

A similar study was conducted by Hours et al. (2003): they analysed 102 male workers employed in 3 French urban incinerators during 1996, matching for age with 94 male workers from other industrial activities. The exposed workers were distributed into 3 categories of exposure based on air sampling at the workplace: crane and equipment operators, furnace workers, and maintenance and effluent-treatment workers. An excess of respiratory problems, mainly daily cough, was more often resulted in the exposed groups, and a significant relationship between exposure and decreases of several pulmonary parameters was also observed, after adjustment for tobacco consumption and centre. For the maintenance and effluent group, as well as for the furnace group, elevated relative risks were estimated for skin symptoms.

In the same year, Takata et al. (2003) conducted a cross-sectional study in Japan on 92 workers of a municipal solid waste incinerator to investigate the health effects of chronic exposure to dioxins. The concentrations of these chemicals among the blood of the workers who had engaged in maintenance of the furnace, the electric dust collector, and the wet scrubber of the incinerator were higher compared with those of residents in surrounding areas, but there were no clinical signs or findings correlated to blood levels of dioxins.

In sum, some suggestions of increased gastric cancer and respiratory problems among incinerators workers are available. There is a very high level of uncertainties to derive conclusions.

Epidemiological studies of health effects of other waste management technologies

Ten epidemiologic studies on the potential adverse health effects of other waste management practices are reviewed and listed in Appendix 4.

<u>Waste collection</u>

Ivens et al. (1997a) investigated the adverse health effects among waste collectors in Denmark. In a questionnaire based survey among 2303 waste collectors and a comparison group of 1430 male municipality workers, information on self-reported health status and working conditions was collected and related to estimated level of bioaerosol exposure. After adjustment for several confounders (average alcohol consumption per day, smoking status, and the psychosocial exposure measures demand and support), a dose-response relationship between level of exposure to fungal spores and self-reported diarrhoea was indicated, meaning that the higher weekly dose, the more reports of gastrointestinal symptoms.

In contrast to these results, a study on 853 workers employed by 27 municipal household waste collection departments in Taiwan did not find an excess of gastrointestinal symptoms (Yang et al., 2001). The workers answered a survey questionnaire and were classified into two occupational groups by specific exposures on the basis of the recorded designation of their specific task. The exposed group included those working in the collection of mixed domestic waste, front runner or loader, collection of separated waste and special kinds of domestic waste (paper, glass, etc.), garden waste, bulky waste for incineration, and the vehicle driver; the control group included accountants, timekeepers, canteen staff, personnel, and other office workers. No significant differences were found in the prevalence of gastrointestinal symptoms, but results indicated that all respiratory symptom prevalence, except dyspnea, were significantly higher in the exposed group, after adjustment for age, sex, education, smoking status, and duration of employment.

Composting facilities

In a German cross sectional study by Bunger et al. (2000), work related health complaints and diseases of 58 compost workers and 53 biowaste collectors were investigated and compared with 40 control subjects. Compost workers had significantly more symptoms and diseases of the skin and the airways than the control subjects, although no correction was performed for the confounding effect of smoking as there was no significant difference in smoking habits between the three groups.

A subsequent study in Germany by Herr et al. (2003) examined the health effects of bioaerosol emitted by a composting plant on community residents. A total of 356 questionnaires from residents living at different distances from a composting site, and from unexposed controls were collected: self reported prevalence of health complaints during the past years, doctors' diagnoses, as well as residential odour annoyance were assessed, and microbiological pollution was measured simultaneously in residential outdoor air. Reports of irritative airway complaints were associated with residency in the highest bioaerosol exposure category, 150-200 m (versus residency >400-500 m) from the site, and period of residency more than five years. No residential odour annoyance was detected.

Materials recycling facilities

There are no epidemiological studies of populations living near materials recycling facilities, only studies on workers are available.

In the already quoted study of Rapiti et al. (1997) on workers at two municipal plants for incinerating and garbage recycling, increased risk was found for gastric cancer in the category with more than 10 years since first exposure, in contrast with reduced mortality from lung cancer.

In the study by Rix et al. (1997), 5377 employees in five paper recycling plants in Denmark between 1965 and 1990 were included in a historical cohort, and the expected number of cancer cases was calculated from national rates. The incidence of lung cancer was slightly increased among men in production and moderately increased in short term workers with less than 1 year of employment; there was significantly more pharyngeal cancer among male, but this increase may be influenced by confounders such as smoking and alcohol intake.

Sigsgaard et al. (1994) conducted a cross-sectional study to examine the workshift changes in lung function among 99 recycling workers (resource recovery and paper mill workers), correlating these findings with measurements of total dust and endotoxins. Exposure to organic dust caused a fall in FEV_1 over the workshift, and this was significantly associated with the exposure to organic dust; no significant association was found between endotoxin exposure and lung function decrements.

The same authors (Sigsgaard et al., 1997) also analysed skin and gastrointestinal symptoms among 40 garbage handling, 8 composting and 20 paper sorting workers from all over Denmark, and an increased risk of itching of the skin and vomiting or diarrhoea in the garbage handling was found.

In a nationwide study, Ivens et al. (1997b) reported findings of self-reported gastrointestinal symptoms by self-reported type of plant. A questionnaire based survey among Danish waste recycling workers at all composting, biogas-producing, and sorting plants collected data on occupational exposures (including questions on type of plant, type of waste), present and past work environment the psychosocial work environment, and health status. Prevalence rate ratios adjusted for other possible types of job and relevant confounders were estimated with a comparison group of non-exposed workers, and an association was found between sorting paper and diarrhoea, between nausea and work at plastic sorting plants, and non significantly between diarrhoea and work at composting plants.

The health status of workers employed in the paper recycling industry was also studied by Zuskin et al. (1998). A group of 101 male paper-recycling workers employed in one paper processing plant in Croatia, and a group of 87 non exposed workers employed in packing food products in the food industry was studied for the prevalence of chronic respiratory symptoms, and results indicated significantly higher prevalences of all chronic respiratory disturbs were found in paper compared with control workers.

More recently, Gladding et al. (2003) studied 159 workers from nine materials recovery facilities (MRFs) in United Kingdom. Measurements of airborne total dust, endotoxin, (1-3)-beta-D-glucan, and a questionnaire survey were carried out. The results suggest that materials recovery facilities workers exposed to higher levels of endotoxin and (1-3)-beta-D-glucan at their work sites experience various work-related symptoms, and that the longer a worker is in the MRF environment, the more likely he is to become affected by various respiratory and gastrointestinal symptoms.

Choosing relative risk estimates for health impact assessment of residence near landfills and incinerators

The reviewed studies have been used to summarize the evidence available, as it is indicated in table 1. Only when the overall degree of evidence was considered at least “limited”, some relative risk estimates have been extracted so that they can be used in the health impact assessment process. Table 2 summarizes the relevant figures for health effects related to landfills and incinerators that are most reliable. For each relative risk the distance from the source has been reported as well as the overall level of scientific uncertainty of the effect estimates based on an arbitrary scale: very high, high, moderate, low, very low.

<u>Landfills</u>

From the review presented above, it is clear that the studies on cancer are not sufficient to draw conclusions regarding a health effect near landfills. The two studies from Goldberg et al (1999) and Pukkala et al (2001) are not consistent with regards to the cancers sites, with the only exception of pancreatic cancer. The largest study conducted in England by Jarup et al (2002) is not suggesting an increase for the cancer sites that were investigated. For other chronic diseases, especially respiratory diseases, investigations are lacking with only one suggestive indication of an increased risk of asthma in adults (Pukkala et al, 2001) but with no replication of the findings. Overall, the evidence that living near landfills may be associated with health effects in adults is inadequate.

A different picture appears for congenital malformations and low birth weight where a limited evidence exists of an increased risk for babies born to mothers living near landfill sites. The relevant results come from Elliott et al. (2001). Statistically significant increased risk were found for all congenital malformations, neural tube defects, abdominal wall defects, surgical correction of gastroschisis and exomphalos, and low and very low birth weight for births occurring in people living within 2 km from the sites. Although several alternative explanations, including ascertainment bias, and residual confounding cannot be excluded in the study, Elliott et (2001) provides quantitative effect estimates whose level of uncertainty can be considered as moderate. In addition, the effect on low birth weight has been confirmed in the complete study from Alaska (Gilbreath et al, 2006).

Incinerators

Quantitative estimates of excess risk of specific cancers in populations living near solid waste incinerator plants were provided by Elliott et al. (1996). We have reported in table 2 the effect estimates for all cancers, stomach, colon, liver, and lung cancer based on their “second stage” analysis. There was an indication of residual confounding from socioeconomic status near the incinerators and a concern of misdiagnosis among registrations and death certificates for liver cancer. The histological review of the liver cancer cases was done, giving a re-estimation of the previously calculated excess risk (from 0.95 excess cases 10^{-5}/year to between 0.53 and 0.78 excess cases 10^{-5}/year). We then score the uncertainty for these tumours as “moderate” with the exception of liver cancer (low) since the reassessment of misdiagnosis was done and the extent of residual confounding was lower. In the Elliott et al (1996) study no significant decline in risk with distance for non-Hodgkin lymphoma and soft tissue sarcoma was found. However, the studies of Viel et al (2000) and Floret (2003) conducted in France and the study from Comba et al (2003) in Italy provide some indications that an excess of these form of cancers may be related to emissions of dioxin from incinerators. In fact, a recent study by Zambon et al. (2007) clearly showed a significant increase in the risk of soft-tissue sarcoma, correlated both with the level and the length of environmental modelled exposure to dioxin-like substances emitted by waste incinerators. (We not included this study in the review because the literature search was limited to 31/12/2006). As a result, we provided effect estimates in table 2 also for non-Hodgkin lymphoma and soft tissue sarcoma as derived from the conservative “first stage” analysis conducted by Elliott et al (1996). We scored the level of uncertainty of this relative risk estimates as “low”.

With regards to congenital malformations near incinerators, Cordier et al (2004) provided effect estimates for facial cleft and renal dysplasia as they were more frequent in the “exposed” communities living within 10 km from the sites. Other reproductive effects, such as an effect on twinning or sex determination, have been described; however the results are inadequate.

Conclusions

We have conducted a systematic review of the literature regarding health effects of waste management. After the extensive review, in many cases the overall evidence was inadequate to establish a relationship between a specific waste process and health effects. However, at least for some associations a limited evidence has been found and few studies were selected for a quantitative evaluation of the health effects. These relative risks could be used for health impact assessment but it should be considered that the level of uncertainty in these effect estimates is at least moderate for most of them.

It is clear that future research into the health risks of waste management needs a more accurate characterization of individual exposure, an improved knowledge of chemical and toxicological data on specific compounds, multi-site studies on large populations to increase statistical power, approaches based on individuals rather than communities and a better control of confounding factors.

Table 1. Summary of the overall epidemiologic evidence on municipal solid waste disposal: landfills and incinerators.

HEALTH EFFECT	LEVEL OF EVIDENCE	
	LANDFILLS	**INCINERATORS**
All cancer	Inadequate	Limited
Stomach cancer	Inadequate	Limited
Colorectal cancer	Inadequate	Limited
Liver cancer	Inadequate	Limited
Larynx cancer	Inadequate	Inadequate
Lung cancer	Inadequate	Limited
Soft tissue sarcoma	Inadequate	Limited
Kidney cancer	Inadequate	Inadequate
Bladder cancer	Inadequate	Inadequate
Non Hodgkin's lymphoma	Inadequate	Limited
Childhood cancer	Inadequate	Inadequate
Total birth defects	Limited	Inadequate
Neural tube defects	Limited	Inadequate
Orofacial birth defects	Inadequate	Limited
Genitourinary birth defects	Limited[2]	Limited[3]
Abdominal wall defects	Inadequate	Inadequate
Gastrointestinal birth defects[4]	Inadequate	Inadequate
Low birth weight	Limited	Inadequate
Respiratory diseases or symptoms	Inadequate	Inadequate

"*Inadequate*": available studies are of insufficient quality, consistency, or statistical power to decide the presence or absence of a causal association. "*Limited*": a positive association has been observed between exposure and disease for which a causal interpretation is considered to be credible, but chance, bias, or confounding could not be ruled out with reasonable confidence.

---

[2] Hypospadias and epispadias
[3] Renal dysplasia
[4] The original estimates were given for "surgical corrections of gastroschisis and exomphalos"

Table 2. Relative risk estimates for community exposure to landfills and incinerators

3.1.1. Outcome	Distance from the source	Relative Risk (Confidence Interval)	Level of confidence[2]
Landfills			
Congenital malformations (Elliott et al, 2001)			
All congenital malformations	Within 2 km	1.02 (99% CI=1.01-1.03)	Moderate
Neural tube defects	Within 2 km	1.06 (99% CI=1.01-1.12)	Moderate
Hypospadias and epispadias	Within 2 km	1.07 (99% CI=1.04-1.11)	Moderate
Abdominal wall defects	Within 2 km	1.05 (99% CI=0.94-1.16)	Moderate
Gastroschisis and exomphalos[1]	Within 2 km	1.18 (99% CI=1.03-1.34)	Moderate
Low birth weight (Elliott et al, 2001)	Within 2 km	1.06 (99% CI=1.052-1.062)	High
Very low birth weight	Within 2 km	1.04 (99% CI=1.03-1.06)	High
Incinerators			
Congenital malformations (Cordier et al, 2004)			
Facial cleft	Within 10 km	1.30 (95% CI=1.06-1.59)	Moderate
Renal dysplasia	Within 10 km	1.55 (95% CI=1.10-2.20)	Moderate
Cancer (Elliott et al, 1996)			
All cancer	Within 3 km	1.035 (95% CI=1.03-1.04)	Moderate
Stomach cancer	Within 3 km	1.07 (95% CI=1.02-1.13)	Moderate
Colorectal cancer	Within 3 km	1.11 (95% CI=1.07-1.15)	Moderate
Liver cancer	Within 3 km	1.29 (95% CI=1.10-1.51)	High
Lung cancer	Within 3 km	1.14 (95% CI=1.11-1.17)	Moderate
Soft-tissue sarcoma	Within 3 km	1.16 (95% CI=0.96-1.41)	High
Non-Hodgkin's lymphoma	Within 3 km	1.11 (95% CI=1.04-1.19)	High

[1] The original estimates were given for "surgical corrections of..". [2] The following scale for the level of confidence has been adopted: very high, high, moderate, low, very low.

References

Altavista P, Belli S, Bianchi F, Binazzi A, Comba P, Del Giudice R, Fazzo L, Felli A, Mastrantonio M, Menegozzo M, Musmeci L, Pizzuti R, Savarese A, Trinca S, Uccelli R. 2004 Cause-specific mortality in an area of Campania with numerous waste disposal sites. *Epidemiologia & Prevenzione*. 28:311-21. Italian

Bianchi F, Minichilli F. 2006 Mortality for non-Hodgkin lymphoma in the period 1981-2000 in 25 Italian municipalities with urban solid waste incinerators. *Epidemiologia & Prevenzione*. 30:80-1. Italian

Biggeri A, Barbone F, Lagazio C, Bovenzi M, Stanta G. 1996 Air pollution and lung cancer in Trieste, Italy: spatial analysis of risk as a function of distance from sources. *Environmental Health Perspectives*. 104:750-4.

Biggeri A, Catelan D. 2005 Mortality for non-Hodgkin lymphoma and soft-tissue sarcoma in the surrounding area of an urban waste incinerator. Campi Bisenzio (Tuscany, Italy) 1981-2001. Epidemiologia & Prevenzione. 29:156-9. Italian

Bresnitz EA, Roseman J, Becker D, Gracely E. 1992 Morbidity among municipal waste incinerator workers. American Journal of Industrial Medicine. 22:363-78.

Bunger J, Antlauf-Lammers M, Schulz TG, Westphal GA, Muller MM, Ruhnau P, Hallier E. 2000 Health complaints and immunological markers of exposure to bioaerosols among biowaste collectors and compost workers. *Occupational Environmental Medicine*. 57:458-64.

Department for Environment, Food and Rural Affairs (DEFRA). 2004 Review of *Environmental and Health Effects of waste management: municipal solid waste and similar wastes*.

Dolk H, Vrijheid M, Armstrong B, Abramsky L, Bianchi F, Garne E, Nelen V, Robert E, Scott JE, Stone D, Tenconi R. 1998 Risk of congenital anomalies near hazardous-waste landfill sites in Europe: the EUROHAZCON study. *Lancet*. 352:423-7

Dolk H, Vrijheid M. 2003 The impact of environmental pollution on congenital anomalies. *British Medical Bulletin*. 68:25-45.

Dummer TJ, Dickinson HO, Parker L. 2003 Adverse pregnancy outcomes around incinerators and crematoriums in Cumbria, north west England, 1956-93. *Journal of Epidemiology Community Health*. 57:456-61.

Elliott P, Shaddick G, Kleinschmidt I, Jolley D, Walls P, Beresford J, Grundy C. 1996 Cancer incidence near municipal solid waste incinerators in Great Britain. *British Journal of Cancer*. 73:702-10.

Elliott P, Eaton N, Shaddick G, Carter R. 2000 Cancer incidence near municipal solid waste incinerators in Great Britain. Part 2: histopathological and case-note review of primary liver cancer cases. *British Journal of Cancer*. 82:1103-6.

Elliott P, Briggs D, Morris S, de Hoogh C, Hurt C, Jensen TK, Maitland I, Richardson S, Wakefield J, Jarup L. 2001 Risk of adverse birth outcomes in populations living near landfill sites. *British Medical Journal*. 323:363-8.

Fielder HM, Poon-King CM, Palmer SR, Moss N, Coleman G. 2000 Assessment of impact on health of residents living near the Nant-y-Gwyddon landfill site: retrospective analysis. *British Medical Journal*. 320:19-22.

Franchini M, Rial M, Buiatti E, Bianchi F. 2004 Health effects of exposure to waste incinerator emissions: a review of epidemiological studies. *Annali dell' Istituto Superiore di Sanità*. 40:101-15. Italian

Fukuda Y, Nakamura K, Takano T. 2003 Dioxins released from incineration plants and mortality from major diseases: an analysis of statistical data by municipalities. *Journal of Medical and Dental Sciences*. 50:249-55.

Gelberg KH. 1997 Health study of New York City Department of Sanitation landfill employees. *Journal of Occupational and Environmental Medicine*. 39:1103-10.

Gladding T, Thorn J, Stott D. 2003 Organic dust exposure and work-related effects among recycling workers. *American Journal of Industrial Medicine*. 43:584-91.

Gustavsson P. 1989 Mortality among workers at a municipal waste incinerator. *American Journal of Industrial Medicine*. 15:245-53.

Herr CE, Zur Nieden A, Jankofsky M, Stilianakis NI, Boedeker RH, Eikmann TF. 2003 Effects of bioaerosol polluted outdoor air on airways of residents: a cross sectional study. *Occupational and Environmental Medicine*. 60:336-42.

Hours M, Anzivino-Viricel L, Maitre A, Perdrix A, Perrodin Y, Charbotel B, Bergeret A. 2003 Morbidity among municipal waste incinerator workers: a cross-sectional study. *International Archive of Occupational and Environmental Health*. 76:467-72.

Hsiue TR, Lee SS, Chen HI. 1991 Effects of air pollution resulting from wire reclamation incineration on pulmonary function in children. *Chest*. 100:698-702.

Hu SW, Shy CM. Health effects of waste incineration: a review of epidemiologic studies. 2001 *Journal of the Air & Waste Management Association*. 51:1100-9.

Ivens UI, Hansen J, Breum NO, Ebbehoj N, Nielsen BH, Poulsen OM, Wurtz H, Skov T. 1997 Diarrhoea among waste collectors associated with bioaerosol exposure. *Ann Agric Environ Med*. a; 4:63-68.

Ivens UI, Ebbehoj N, Poulsen OM, Skov T. Gastrointestinal symptoms among waste recycling workers. *Ann Agric Environ Med*. 1997 b; 4:153-157.

Jansson B, Voog L. 1989 Dioxin from Swedish municipal incinerators and the occurrence of cleft lip and palate malformations. *Int J Environ Stud*. 34:99-104.

Jarup L, Briggs D, de Hoogh C, Morris S, Hurt C, Lewin A, Maitland I, Richardson S, Wakefield J, Elliott P. 2002 Cancer risks in populations living near landfill sites in Great Britain. *British Journal of Cancer*. 86:1732-6.

Kloppenborg SCh, Brandt UK, Gulis G, Ejstrud B. 2005 Risk of congenital anomalies in the vicinity of waste landfills in Denmark; an epidemiological study using GIS. *Central European Journal of Public Health*. 13:137-43.

Knox E. 2000 Childhood cancers, birthplaces, incinerators and landfill sites. *International Journal of Epidemiology*. 29:391-7.

Lee JT, Shy CM. 1999 Respiratory function as measured by peak expiratory flow rate and PM10: six communities study. *Journal of Exposure Analysis and Environmental Epidemiology*. 9:293-9.

Lloyd OL, Lloyd MM, Williams FL, Lawson A. 1988 Twinning in human populations and in cattle exposed to air pollution from incinerators. *British Journal of Industrial Medicine*. 45:556-60.

Michelozzi P, Fusco D, Forastiere F, Ancona C, Dell'Orco V, Perucci CA. 1998 Small area study of mortality among people living near multiple sources of air pollution. *Occupational and Environonmental Medicine*. 55:611-5.

Minichilli F, Bartolacci S, Buiatti E, Pallante V, Scala D, Bianchi F. 2005 A study on mortality around six municipal solid waste landfills in Tuscany Region. *Epidemiologia & Prevenzione*. 29 S:53-6.

Miyake Y, Yura A, Misaki H, Ikeda Y, Usui T, Iki M, Shimizu T. 2005 Relationship between distance of schools from the nearest municipal waste incineration plant and child health in Japan. *European Journal of Epidemiology*. 20:1023-9.

Palmer SR, Dunstan FD, Fielder H, Fone DL, Higgs G, Senior ML. 2005 Risk of congenital anomalies after the opening of landfill sites. *Environmental Health Perspectives*. 113:1362-5.

Poulsen OM, Breum NO, Ebbehoj N, Hansen AM, Ivens UI, van Lelieveld D, Malmros P, Matthiasen L, Nielsen BH, Nielsen EM, et al. 1995 Sorting and recycling of domestic waste. Review of occupational health problems and their possible causes. *The Science of the Total Environment*. 168:33-56.

Poulsen OM, Breum NO, Ebbehoj N, Hansen AM, Ivens UI, van Lelieveld D, Malmros P, Matthiasen L, Nielsen BH, Nielsen EM, et al. Collection of domestic waste. 1995 Review of occupational health problems and their possible causes. *The Science of the Total Environment*. 170:1-19.

Parodi S, Baldi R, Benco C, Franchini M, Garrone E, Vercelli M, Pensa F, Puntoni R, Fontana V. 2004 Lung cancer mortality in a district of La Spezia (Italy) exposed to air pollution from industrial plants. *Tumori*. 90:181-5.

Rapiti E, Sperati A, Fano V, Dell'Orco V, Forastiere F. 1997 Mortality among workers at municipal waste incinerators in Rome: a retrospective cohort study. *American Journal of Industrial Medicine*. 31:659-61.

Rix BA, Villadsen E, Engholm G, Lynge E. 1997 Risk of cancer among paper recycling workers. *Occupational and Environmental Medicine*. 54:729-33.

Rushton L. 2003 Health hazards and waste management. *British Medical Bulletin*. 68:183-97.

Rydhstroem H. 1998 No obvious spatial clustering of twin births in Sweden between 1973 and 1990. *Environmental Research*. 76:27-31.

Shy CM, Degnan D, Fox DL, Mukerjee S, Hazucha MJ, Boehlecke BA, Rothenbacher D, Briggs PM, Devlin RB, Wallace DD, et al. 1995 Do waste incinerators induce adverse respiratory effects? An air quality and epidemiological study of six communities. *Environmental Health Perspectives*. 103:714-24.

Sigsgaard T, Abel A, Donbaek L, Malmros P. 1994 Lung function changes among recycling workers exposed to organic dust. *American Journal of Industrial Medicine*. 25:69-72.

Sigsgaard T, Hansen J, Malmros P. Biomonitoring and work related symptoms among garbage handling workers. Ann Agric Environ Med. 1997; 4:107-112.

Takata T. 2003 Survey on the health effects of chronic exposure to dioxins and its accumulation on workers of a municipal solid waste incinerator, rural part of Osaka Prefecture, and the results of extended survey afterwards. *Industrial Health*. 41:189-96.

Tango T, Fujita T, Tanihata T, Minowa M, Doi Y, Kato N, Kunikane S, Uchiyama I, Tanaka M, Uehata T. 2004 Risk of adverse reproductive outcomes associated with proximity to municipal solid waste incinerators with high dioxin emission levels in Japan. *Journal of Epidemiology*. 14:83-93.

Vrijheid M. 2000 Health effects of residence near hazardous waste landfill sites: a review of epidemiologic literature. *Environmental Health Perspectives*. 108 S:101-12.

World Health Organization. 2007 Population health and waste management: scientific data and policy options. Report of a WHO workshop Rome, Italy, 29–30 March 2007. WHO, Regional Office for Europe, Copenhagen.

Williams FL, Lawson AB, Lloyd OL. 1992 Low sex ratios of births in areas at risk from air pollution from incinerators, as shown by geographical analysis and 3-dimensional mapping. *International Journal of Epidemiology*. 21:311-9.

Yang CY, Chang WT, Chuang HY, Tsai SS, Wu TN, Sung FC. 2001 Adverse health effects among household waste collectors in Taiwan. *Environmental Research*. 85:195-9.

Zuskin E, Mustajbegovic J, Schachter EN, Kanceljak B, Kern J, Macan J, Ebling Z. 1998 Respiratory function and immunological status in paper-recycling workers. J *Occup Environ Med*. 40:986-93.

APPENDIX 7

Estimating attributable cancer incidence around incinerators

Rationale

1. The basic formula to compute the number of cancer cases attributable to an incinerator is:

   $AC = Rate_{unex} * ER * Pop_{exp}$

   where AC = the attributable cancer incidence

   $Rate_{unex}$ = background incidence rate in the general population

   ER = excess risk in the exposed population (relative risk – 1)

   Pop_{exp} = number of exposed people
2. We have relative risks calculated only for an arbitrarily defined exposed population (e.g. in terms of distance from an incinerator, Elliott et al.1996). Although the possibility to inference causality from these studies is limited (due to limitations of the studies discussed above), these estimates are the unique starting point for our assessment.
3. Once we have assumed that there is a relationship between living near incinerators and cancer incidence, we may suspect that the excess risk is not constant over time, but varies for a specific individual of the population at a give age and specific time as a function of various characteristics: level of attained cumulative exposure, latency since first exposure and latency since cessation of exposure (if any).
4. We therefore need to assume a theoretical model of cancer occurrence and to impute the varying excess risk around different incinerators, as a function of the different characteristic of the plant and of the nearby population.

Assumptions

1. <u>Model of carcinogenesis.</u> We do not have clear scientific data about the carcinogenic model underlying the association between living close to the plants and occurrence of cancer. We may assume here that the model that better fit our purpose is the most studies one that relates cigarette smoking with lung cancer. Under the multistage theory of cancer proposed by Armitage and Doll (Armitage and Doll, 1958), Doll and Peto (1978) indicated that the excess relative risk of lung cancer is a function of attained age together with a complex dependency related to age at starting, duration and intensity of smoking and time since quitting. Various attempts have been made to validate the model using data from real long term cohorts (Hazelton et al. 2005; Schollnberger et al. 2006). Although the results of these studies do not provide a uniform response regarding the role of each factor (Hornsby et al. 2007), and the approach may be seen as a simplification, it has the advantage to provide a template for addressing other exposure-response relationships (Siemiatycki, 2005). It is clear that this model that is mostly applicable to solid cancers of epithelial origin. The approach could be different for hematological or soft tissues cancers or for childhood cancer. Finally, the model that we assume is multiplicative in nature, namely that the excess risk is a multiplicative function of the baseline risk.
2. <u>Uniform excess risk in the area within 3 Km.</u> We may assume that on a given year the excess risk cross all exposed areas around a given incinerator in the study (3 km) is equal to that derived from the scientific literature with corrections depending on several factors referenced above.
3. <u>Reference Excess Risk (RER).</u> We may assume as reference that the value of 3.5% (95%CI: 3-4%) excess risk reported in the paper by Elliott et al (1996) reflects the additional risk of total cancer incidence for a population living within 3 km from an

incinerator exposed for a duration of 20 years at the levels of contamination that were present in the period 1960-1980. We can call this value Reference Excess Risk (RER). In fact, all the 72 incinerators studied in the Elliott's paper did start operation before 1976, the follow-up was conducted during 1974-1986 (1974-1987 for Wales and 1975-1987 for Scotland), and the effect estimate was given considering 10 years of latency for solid cancers.

4. Exposure levels vary with time. We may assume that in subsequent years after 1980, due to technological improvements and as results of national and European laws, the emissions from incinerators have been reduced. For instance, measured particulate matter emissions from one incinerator in Italy (Modena) where 0.19 g/s in 1980-1989 (two lines), 0.0347 and 0.376 g/s in 1995-1996 (two lines), 0.0196, 0.0273 and 0.104 g/s (three lines) in 1997-2002, and 0.0081, 0.0101, and 0.013 g/s (three lines) in 2003-2006. On the other hand, emission limits in the UK were reduced through legislation from 460 mg/m3 (1968) to 200 mg/m3 (1983) to 30 mg/m3 (1989/1990) and finally to 10 mg/m3 in 2000. On the basis of these data, we can assume that if the exposure level was 1 before 1980, it was 0.8 in 1980-1989, 0.2 in 1990-2000, and 0.05 after 2000. In other words, we are assuming that the exposure levels during the eighties were lower (0.8) that during the seventies, during the nineties were fourfold lower, and in more recent times they were twentyfold lower that the seventies. Of course, these assumptions may be varied in sensitivity analysis.
5. Calculation of cumulative exposure. We need to recognise that at a given age of a person, the best way to summarize the exposure experience is to calculate cumulative exposure (CE) as the sum of the exposure contribution during the different periods. The analogues for cigarette smoking are pack-years. For example, a person aged 60 in 2001, living nearby an incinerator opened in 1980 and still running in 2001, will have over the period 1980-2001 a CE of 10.25 (8+2.2+0.05=10.25, i.e. 10 years at exposure 0.8 in 1980-1989, 11 years at exposure 0.2 in 1990-2000, and one year in 2001 at exposure 0.05).
6. Latency since first exposure and latency since exposure cessation. Finally, latency since first exposure is a relevant issue, especially if a long time for the evaluation is to be considered. For most solid cancers, there is some cancer expression only several years after first exposure to carcinogens and the full effect is appreciable only after 20 years (as indicated above, latency may be shorter for non solid cancers). In our case, we assume that the effect of the exposure to a given incinerator will be appreciable only after some years from first exposure, the peak will be reached after 20 years and it will be constant up to 40 years, then it will start to smoothly decline approaching 0 after 70-80 years. On the other hand, if the exposure is removed, as in the case of smoking cessation, the risk declines as a function of the time since cessation. We may assume that the excess risk will smoothly decline soon after cessation of exposure.
7. For practical reasons, we need to assume that the population selected on a given year has been always living close to the plant and it size and age composition will be constant during the period of the evaluation.

Calculations

1. Time and age. For a specific age class (*a_i*) of the population we wish to consider, we define the time elapsed (t_{exp}) from the start of exposure to the incinerator (ys) and the reference year (or year of calculation).

$$t_{\exp_{a\_i}} = \min\left\{ \frac{a(M)_i - a(m)_i}{2} ; y - ys \right\} \quad (1),$$

where:

a_i=i-th age class

a(M)i = max. age in i-th class

a(M)i = min. age in i-th class

y=reference year (or year of calculation)

ys= year of start

Example: incinerator Modena (start in 1980), reference year : 2001, age class: 30-34 years

Then $t_{\exp_{a\_30-34}} = \min\left\{\frac{34+30}{2}; 2001\text{-}1980\right\} = \min\{32;20\} = 20$

2. <u>Cumulative exposure</u>. For a given age class, cumulative exposure is given by the following formula:

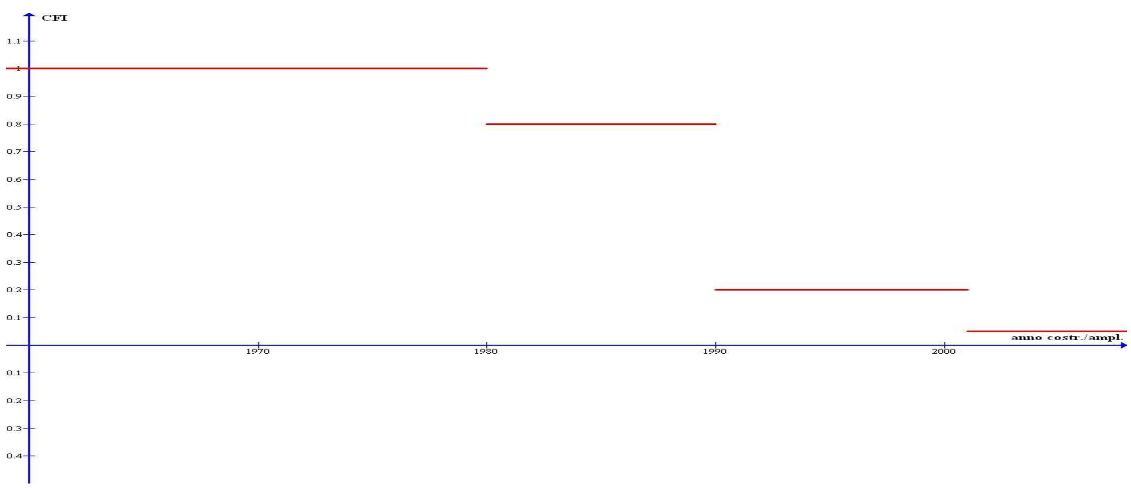
$$CE_{t_{\exp_{a\_i}}} = \sum_{t=1}^{t_{\exp_{a\_i}}} Ey(t) \qquad (2),$$

where *Ey* is the exposure factor for a given year according to the rule:

:

$$Ey(t) = \begin{cases} 1 & ,\ \ t < 1980 \\ 0.8 & ,1980 \le t < 1990 \\ 0.2 & ,1990 \le t < 2001 \\ 0.05 & ,t \ge 2001 \end{cases} \qquad (3)$$

And shown in the graph below.



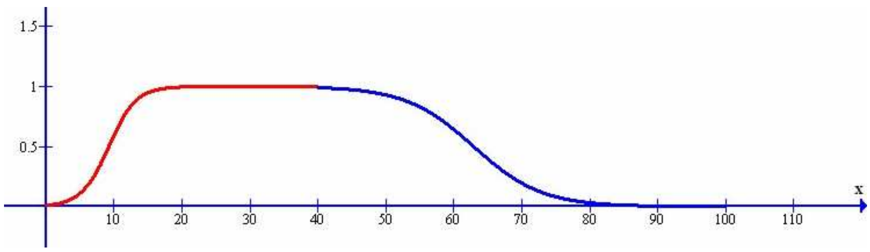
3. Latency since first exposure. We define then latency since start of exposure (Ls) for a given age class (a_*i*) as a function of the time variable indicated above:

$$Ls = f(t_{\exp_{a\_i}}) = \begin{cases} \dfrac{1}{1+b*e^{-ct_{\exp_{a\_i}}}} \quad , \quad t_{\exp_{a\_i}} \leq 40 \\ 1-\dfrac{1}{1+b*e^{-c(t_{\exp_{a\_i}}-40)}} \quad , \quad t_{\exp_{a\_i}} > 40 \end{cases} \qquad (4),$$

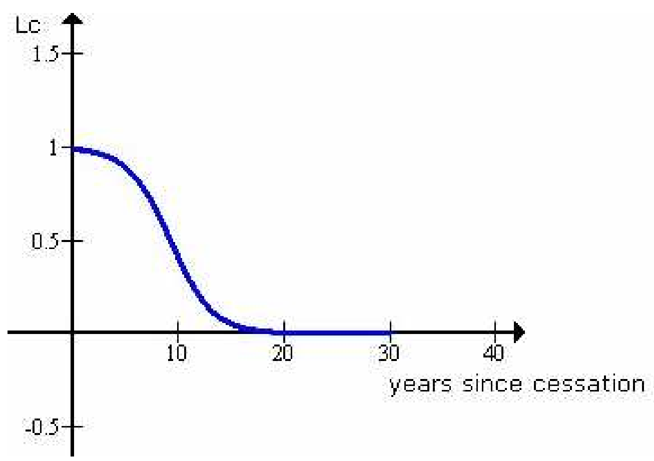
where:

Ls= latency since first exposure

b and c= coefficients for a sigmoid curve that reaches the plateau (one) 20 years since first exposure, remains stable until 40 years, and then starts to decline reaching 0 after 80 years as indicated in the graph below.



4. Latency since cessation of exposure. To allow for the possible effect of cessation of exposure, we assume a factor for latency since cessation of exposure (Lc) that follows a sigmoid with a decrease of the risk starting after the closure and reaching a plateau after 20 years as indicated in the graph below.



This factor follows the function below:

$$Lc = f(t_c) = 1 - \frac{1}{1 + b * e^{-c(t_c)}} \quad (5),$$

Where *tc is time since cessation of exposure*

For each age class and at a given time (year), the three factors indicated above (CE, Ls and Lc) act in a multiplicative way to modify the Reference Excess Risk (from Elliott et al. 1996).

Thus, for a given age class (*a_i*):

$$ER_{a\_i} = RER * (CE_{a\_i} / 20) * Ls * Lc \quad (6),$$

Where

ER_{a_i} = the estimated excess risk of cancer incidence

RER =the reference excess risk as estimate from Elliott et al (1996) (3.5% increase for exposure of 20 years to incinerators operating before 1980).

CE_{a_i} = cumulative exposure

Ls =latency since start of exposure

Lc =latency since cessation of exposure r

Finally, for a given age class (*a_i):*

$$AC_{a\_i} = ER_{a\_i} * Rate_{un\exp} * Pop_{\exp} \quad (7),$$

where

ACa_i =attributable cancer incidence

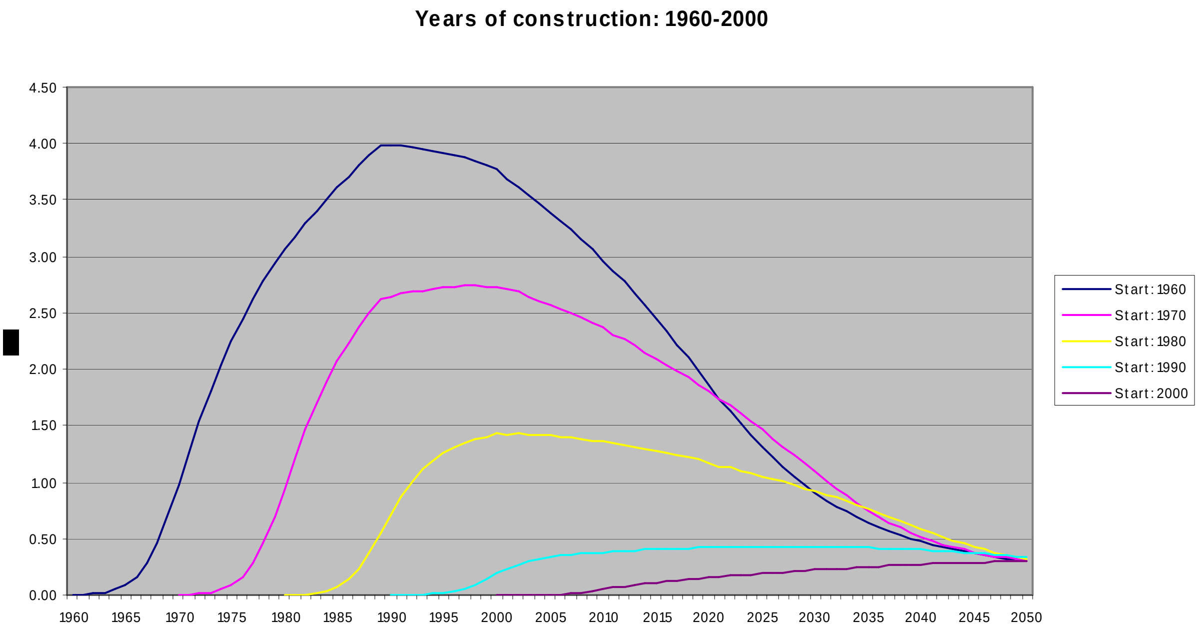
ER a_i =excess risk of cancer incidence

$Rate_{unex}$ =background incidence rate in the general population

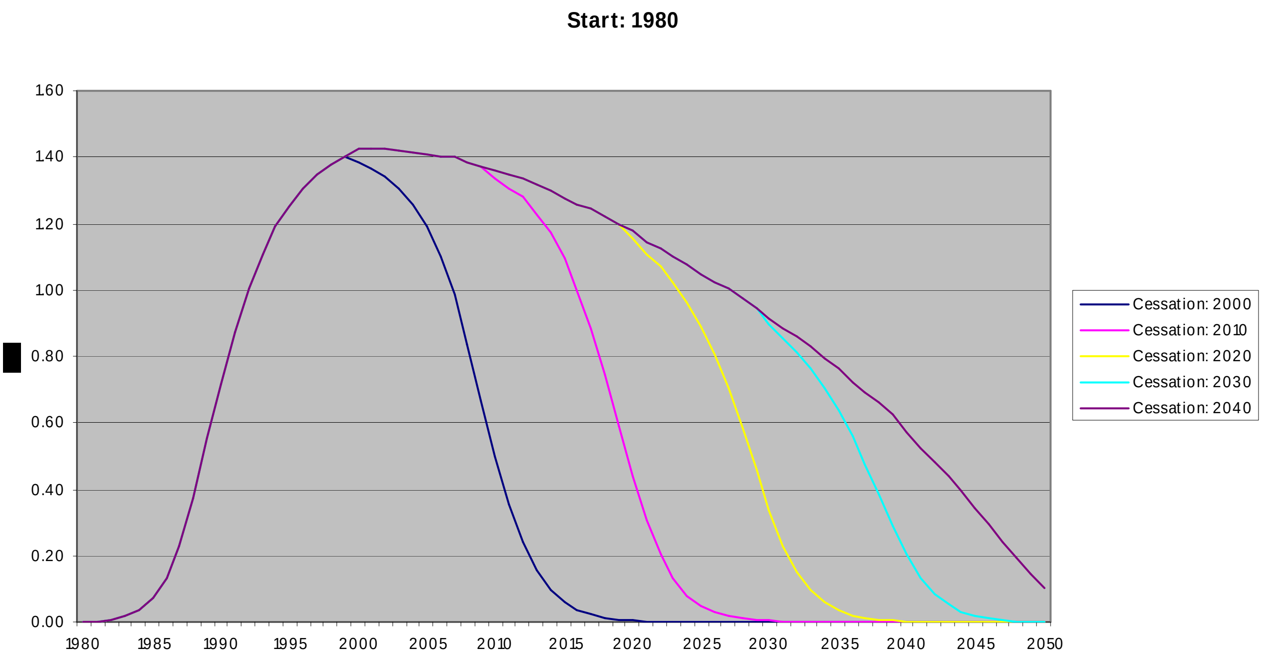
Pop_{exp} =number of exposed people

Results

The figure below shows the results of the application of the model from 1960 to 2050. For each year, the excess risk (ER) (age weighted) of cancer is calculated with reference to a theoretical Italian population (age distribution) living close to an incinerator as function of year of starting operation.



The next figure illustrate the estimated excess risk for a population living close to a plant operating since 1980 as function of the year of closing. The excess risks are reported up to 2050.



References

Armitage P, Doll R. 1954 The age distribution of cancer and a multi-stage theory of carcinogenesis. *British Journal of Cancer*. 8:1-12.

Doll R, Peto R. 1978 Cigarette smoking and bronchial carcinoma: dose and time relationships among regular smokers and lifelong non-smokers. *Journal of Epidemiology and Community Health*. 32:303-13.

Elliott P, Shaddick G, Kleinschmidt I, Jolley D, Walls P, Beresford J, Grundy. 1996 Cancer incidence near municipal solid waste incinerators in Great Britain.*British Journal of Cancer*. 73:702-10.

Hazelton WD, Clements MS, Moolgavkar SH. 2005 Multistage carcinogenesis and lung cancer mortality in three cohorts. *Cancer Epidemiology Biomarkers & Prevention*. 14:1171-81.

Hornsby C, Page KM, Tomlinson IP. 2007 What can we learn from the population incidence of cancer? Armitage and Doll revisited. *Lancet Oncolology*. 8:1030-8.

Schöllnberger H, Manuguerra M, Bijwaard H, Boshuizen H, Altenburg HP, Rispens SM, Brugmans MJ, Vineis P. 2006 Analysis of epidemiological cohort data on smoking effects and lung cancer with a multi-stage cancer model. *Carcinogenesis*. 27:1432-44.

Siemiatycki J. 2005 Synthesizing the lifetime history of smoking. *Cancer Epidemiology Biomarkers & Prevention*. 14:2294-5.