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**POPULATION HEALTH AND WASTE MANAGEMENT:
SCIENTIFIC DATA AND POLICY OPTIONS**

Available evidence on the health effects of environmental exposures from waste incinerators and landfills was reviewed and discussed in a WHO workshop, attended by a group of international experts in epidemiology, environmental science, public health and economics, together with representatives of interest groups. The implications of such evidence were discussed in terms of policy action on waste management in the European context, with special emphasis on the need of limiting and removing harmful exposures and ensuring healthy environmental conditions.

Limitations and uncertainties in available science, deriving mainly from study design and exposure characterization, were described, with the aim of identifying knowledge gaps and priority needs in research.

Sessions were also dedicated to European case studies on health effects of landfills and incinerators, to economic evaluations of waste management options, and to methods and applications of participatory approaches for developing health-friendly policy response to the growing challenge of waste management in Europe.

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Population health and waste management: scientific data and policy options

Report of a WHO workshop
Rome, Italy, 29-30 March 2007



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ABSTRACT

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Executive summary

Waste management is becoming an increasingly complex matter in many European countries. Effects on health and well-being of human exposure both to waste materials and to the products of waste management are, in several instances in Europe, a cause of concern. The Regional Office for Europe of the World Health Organization organized an expert workshop to review the available evidence on health effects and exposures involved in waste landfilling and incineration, analyse key European case studies and discuss how to support European authorities at various level on waste policy-making, by taking into account the health considerations. This report illustrates the findings of the workshop.

With regards to waste landfills, a wide variety of exposures, exposure pathways and exposure scenarios are involved, entailing a large complexity and difficulty in estimating the health risks possibly involved. Only few epidemiological studies have evaluated sites with respect to the types of chemicals they contain and release; most studies on the health effects of waste landfills in fact lack direct exposure measurement, and rely on residential distance from the site or sometimes on exposure modelling. Many health endpoints have been considered in epidemiological studies, including cancer incidence and mortality and reproductive outcomes such as birth defects and low birth weight. Despite the methodological limitations, the scientific literature on the health effects of landfills provides some indication of the association between residing near a landfill site and adverse health effects. The evidence, somewhat stronger for reproductive outcomes than for cancer, is not sufficient to establish the causality of the association. However, in consideration of the large proportion of population potentially exposed to landfills in many European countries and of the low power of the studies to find a real risk, the potential health implications cannot be dismissed.

Incinerators have been operating in many European countries since the 1960s and their technology has evolved over time, in general with a reduction of emissions to nearby communities. As to the possible health effects of incinerators, reasons for concern are inhalation of airborne pollutants resulting from combustion and from incomplete combustion, consumption of contaminated foods and water, or contact with contaminated soil. Information on the presence of hazardous agents in the vicinity of an incinerator is not easily translated into useful exposure measures. Compared to landfills, fewer epidemiological studies are available. While some positive studies exist, the evidence is, overall, not conclusive to establish the occurrence and magnitude of risks. As in landfill studies, increases in relative risk are difficult to detect because they are generally caused by long-term low-level exposures. Studies pointing to an increase in soft tissue sarcomas (STS) and non Hodgkin's lymphomas (NHL) support a possible etiologic role of 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8 TCDD). The evidence is inadequate to draw conclusions that can be used to determine optimal policy choices on incineration: relatively few good quality studies exist and they refer mostly to old generation incineration plants – an important distinction, as stack emissions from modern plants are much reduced compared to old generation plants. The adoption of emission-abating technology, enforced by the European Union (EU), has resulted in a less likely occurrence of measurable health effects on populations resident in the proximity of new generation incinerators. However their overall impact on the general environment and on human health through indirect mechanisms of action has not yet been evaluated. In particular waste incineration, currently on the increase in many countries, may be a non-negligible contributor of greenhouse gases and persistent pollutants on a global scale.

Further insights on health effects of landfills and incinerators are likely to be gained only from studies that consider exposure pathways and biomarkers of exposure and effect, and compare waste-related exposures with those due to other sources of pollution.

The evidence of adverse health effects related to landfills and incinerators, although not conclusive, adds to other environmental concerns in directing waste management strategic choices towards reduction of waste production, re-use and recycling schemes, as prescribed by EU Directives. National and local authorities should oppose and eliminate poor, outdated and illegal practices of waste disposal, which still affect some local communities, support regulation and enforcement, and invest in state-of-the-art technology for lowering emissions.

The decision-making process concerning location and operation of waste facilities should be transparent and fair, and aim at replacing poor or even illegal waste management practices to legal and safe operations, avoiding long delays.

The decision to adopt epidemiological surveillance programs in areas impacted by landfills or incinerators should be taken on the basis of a feasibility analysis aimed at avoiding the execution of non-informative studies. In the cases in which epidemiological surveillance appears to be appropriate, suitable protocols should be adopted, including on evaluation research after major interventions.

Priority needs for research include development and application of biomonitoring, both in human observational studies and in toxicological research, the use of pharmacokinetic models to assess the influence of factors such as metabolism and timing of exposures, and the analysis of all relevant environmental matrices, in order to evaluate chemical exposure pathways and to assess the exposure for specific subsets of the population.

Regardless of the final decision on the appropriateness of a local epidemiological study, actions aimed at addressing a population's concerns should be considered and adopted where necessary, namely:

- provide information on technological standards and process characteristics and environmental mitigation strategies. Resources should also be concentrated on establishing the real level of risk associated with sites, including improved understanding of exposure pathways, before considering site specific epidemiological research;
- provide information on environmental monitoring;
- develop monitoring programs where applicable; and
- enhance communication and participatory activities in order to promote community autonomy and build consensus.

It is also important that the adverse effects on health due to nuisance (smell, noise, litter, effect on property values, stress for lack of regulatory response etc) are considered. These endpoints often escape formal epidemiological analysis but are relevant for the health of communities. Consideration of all relevant health elements may be achieved through integrated and participatory approaches, such as health impact assessment (HIA), which has proven effective in some cases in waste management policies. HIA can be done at a policy, program or project level, and help judge the potential effects of a proposal as well as the distribution of those effects. Understanding and managing the potential or likely health impacts of waste management is likely to be best addressed through either HIA or strategic environmental assessment (SEA). In view of the various limitations hampering our ability to characterize all risks, such assessments should be inspired by a precautionary approach, with respect both to the creation of new facilities and the mitigation of exposure to emissions and leachates of existing sites. In the case of

remediation schemes of existing contaminated sites, priorities should be based on hazard detection, estimation of the size of the exposed population (including vulnerable groups) and appreciation of inequity in the distribution of exposure among population subgroups.

Contributors and editors

This report was produced as a result of the discussions held during the workshop and reflects the contribution of all the participants. Individual presentations are included in Annex A.

The report was edited by Francesco Mitis and Marco Martuzzi (World Health Organization, Regional Office for Europe).

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List of abbreviations

2,3,7,8 T ₄ CDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
8-OH-dG	8-hydroxy-2'-deoxy-guanosine
As	Arsenic
APAT	Italian Agency for the Protection of the Environment and for technical Services
ARPAB	Regional Agency for Environmental Protection of Basilicata Region
ARPA Piedmont	Regional Agency for Environmental Protection of Piedmont Region
ATSDR	The United States Agency for Toxic Substance and Disease Registry
BAT	Best available techniques
BMRs	Hierarchical Bayesian estimators
BRef	BAT Reference
Cd	Cadmium
CH ₄	Methane
CI	Confidence interval
CNCT	National Institute for Occupational Safety and Health
Co	Cobalt
CO	Carbon oxide
CO ₂	Carbon dioxide
Cr	Chromium
Cu	Copper
DALYs	Disability-adjusted life-years
DEFRA	British Department for Environment Food and Rural Affairs
deNO _x	NO _x reduction
EA	Environment Agency
ECRHS	European Community Respiratory Health Survey
EIA	Environmental impact assessment
ENEA	Italian National Agency for New Technologies, Energy and the Environment
ETS	Environmental tobacco smoke
EUROHAZCON	European Collaborative Study of Residence near Hazardous Waste Landfill Sites and Risk of Congenital Malformations
EU	European Union
ExternE	Externalities of Energy
FFQ	Food frequency questionnaire
GIS	Geographical information systems
GPS	Global positioning system
HBM	Human biomonitoring
HC	Hydrocarbons
HCl	Hydrogen chloride
HF	Hafnium
Hg	Mercury
HIA	Health impact assessment
HNO ₃	Nitric acid
ICD IX	International Classification of Disease, ninth revision
INTARESE	Integrated Assessment of Health Risks of Environmental Stressors in Europe
IPC	Integrated Pollution Control
IPPC	Integrated Pollution Prevention and Control
ISS	Istituto Superiore di Sanità
I-TEQ	International toxic equivalent
LCA	Life cycle assessment
LFG	Landfill gas
Mn	Manganese
MSW	Municipal solid waste
MSWI	Municipal solid waste incinerator
NH ₃	Ammonia
NHL	Non Hodgkin's lymphoma
Ni	Nickel
NO ₂	Nitrogen dioxide
NO _x	Nitrogen oxides
NGOs	Nongovernmental organizations
Nimby	Not in my backyard
NMVO	Non-methane volatile organic compounds
OR	Odds ratio
PAH	Polycyclic aromatic hydrocarbons
Pb	Lead

PCB	Polychlorinated biphenyl
PCDD	Polychlorinated dibenzo-p-dioxin
PCDF	Polychlorinated dibenzofuran
PM _{2.5}	Particulate matter with an aerodynamic diameter smaller than 2.5 microns
PM ₅	Particulate matter with an aerodynamic diameter smaller than 5 microns
PM ₁₀	Particulate matter with an aerodynamic diameter smaller than 10 microns
ProVEpA	Environmental Health Surveillance Program
RDF	Refuse-derived fuel
RR	Relative risk
SAHSU	Small Area Health Statistics Unit
Sb	Antimony
SEA	Strategic environmental assessment
SMRs	Standardized mortality ratios
SWI	Solid waste incinerator
SO _x	Sulphur oxides
STS	Soft tissue sarcoma
TEQ	Toxic equivalent
Tl	Thallium
TOC	Total organic compound
V	Vanadium
VOC	Volatile organic compound

1. Introduction

Waste management is becoming an increasingly complex matter in many European countries. Improvements in technology and recycling schemes often are not sufficient to counter growing waste production, obsolescence of existing waste management facilities and shortage of space for the construction of new facilities. Further difficulties are posed, in many instances, by mounting public concerns on effects on health and well-being.

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.

In the last ten years an expert meeting (World Health Organization - Regional Office for Europe, 1998) and a Workshop on the Health Effects of Waste Landfills have been organized by WHO (World Health Organization - European Centre for Environment and Health, 2000). Since 2000, studies have been growing and more sophisticated statistical techniques have been applied, however, problems on the quantification of exposure, assessment of health impacts and interpretation of studies remain and uncertainty is large. It is in fact still difficult to perform a reliable risk assessment exercise.

A second WHO workshop was thus organized in Rome on 29–30 March 2007; the workshop also dealt with the health effects of waste incineration, not addressed by previous WHO reviews. The working group was formed by 26 experts from 11 European countries, working in relevant research disciplines, including environmental epidemiology, health impact assessment, cost-benefit analyses, chemical hazard, public health, preventive medicine and by 19 observers. The programme of the workshop and the full list of participants are attached in Annex B and Annex C.

The workshop aimed at discussing the open questions in waste and its health effects and had the following objectives:

- review the current scientific evidence on health effects of environmental and occupational exposure to waste and waste treatment emissions, with special reference to landfilling and incineration;
- review the contribution of state-of-the-art technology in reducing emissions and population exposures;
- clarify what exposures are likely to be most relevant in terms of health and the degree of population exposures in European countries;
- analyse case studies from European countries where waste management and health is a matter of concern or where successful policies are adopted (comparing and contrasting success stories and unsolved problems);

- review and evaluate health-based approaches to support decision-making in waste management, for example, burden of disease estimation, HIA, treatment of uncertainty, economic evaluations; and
- provide guidance and support to policy-makers engaged in waste management.

The workshop was organized with presentations on the current scientific evidence, given by experts in the field and followed by questions and discussion. This part of the programme was designed to provide scientists with an updated review of the current scientific evidence in the field, on health effects and technological developments of landfills and incinerators. In addition to this review, the workshop discussed policy options in waste management, to compare and contrast real cases and to allow interested parties to take the floor and present their cases and questions. These contributions aimed at providing public health and environmental health practitioners and people involved in decision-making in waste management, and other interested parties, with a critical evaluation of the implications of the available evidence for developing policies in waste management.

This report summarizes the general discussion and conclusions of the workshop. The document is structured in two sections. In the first section technology, emissions and scientific evidence on health effects attributable to landfills and incinerators are described together with the overall assessment of health impact of waste, the knowledge gaps, uncertainty issues, research priorities and policy options. In the second section most of the presentations given during the workshop are summarized in extended abstracts (Annex A), prepared by the presenting authors.

2. Landfills

2.1 Emissions and exposure

Landfills are complex settings that may determine a wide variety of exposures and exposure scenarios (Table 1), involving a multiplicity of agents with different and partly unknown toxicological profiles. The site factors affecting the likelihood that a landfill leads to potentially harmful population exposure are manifold and can be summarized in its engineering and containment, hydrogeology and topography, the type and quantity of waste contained, the mixing of contents, the presence and depth of leachate and the management practices.

Table 1. Routes of exposure in the nearby residents and in the general population

TARGET POPULATION	POTENTIAL ROUTES OF EXPOSURE
Nearby residents	Inhalation of gases or particles adhered to dust emitted from site
	Ingestion of home grown food contaminated via air, water, soil
	Ingestion of drinking-water obtained from private wells contaminated by leachates
	Bathing and washing in contaminated water–skin contact or inhalation of evaporated volatile organic compounds
	Inhalation of contaminated indoor air via soil, leachates, gas migration
	Skin contact with contaminated soil, or inhalation of evaporation from soil
General population	Recreational use of areas close to landfill or contaminated land
	Ingestion of contaminated agricultural produce
	Drinking water from contaminated municipal supplies

The main concerns on health consequences derive from emissions of chemical mixtures or infectious agents with unknown health effects. This problem is amplified in situations where there is limited information and control on landfilling operations, such as with illegal dumping of unknown type and quantity (see, for example, Annex A18).

Epidemiological studies on the health effects of waste landfills exist, but many share the important weakness of the lack of direct exposure measurement: only few sites have been evaluated with respect to the types of chemicals they contain and the extent to which they may be releasing chemicals (Vrijheid, 2000). For this reason the exposure pathways are either modelled (for example using geographical information systems) or, more frequently, assessed through surrogate measures, such as the distance of the residence from the landfill sites.

It must be stressed that, although hypothetical risks can be small and thus difficult to detect against a background of high random variability (given the need to analyse relatively small area units), the potentially exposed population is often large. For instance, in the United Kingdom 80% of the population lives within two kilometres from known landfill sites, hence the overall health impact may be of significant magnitude.

2.2 Scientific evidence

This section describes the scientific evidence on health effects of landfills as summarized by several reviews, published in the last few years.

Fifty papers, reports and abstracts published from 1980 to 1998 were reviewed by Vrijheid (2000). The term landfill was used for any controlled or uncontrolled disposal of waste to land. The results indicated that increases in risk of adverse health effects (low birth weight, birth defects and certain types of cancers) and an increased prevalence of self-reported health symptoms such as fatigue, sleepiness and headaches have been reported near both individual

landfill sites and in some multi-site studies. The author concluded that although a substantial number of studies were conducted, risks to health from landfill sites were hard to quantify because of lack of direct exposure measurement and of the role of confounding factors and suggested more interdisciplinary research to improve levels of knowledge on risks to human health of waste disposal in landfill sites. This should include epidemiological and toxicological studies on individual chemicals and chemical mixtures, well-designed single- and multi-site landfill studies, development of biomarkers and research on risk perception and sociologic determinants of ill health.

In 2002 a review pertaining only to controlled landfills was carried out by Redfearn and Roberts (2002). Single-site studies dealing with unauthorized and unregulated dumps and with sites different from landfills (for example heavy metals mining waste sites or surface impoundments receiving liquid industrial wastes) and multi-sites studies containing a miscellaneous assortment of sites not limited to landfills were, for this reason, excluded from the review. Applying these criteria 13 single-site and 5 multi-site studies were analyzed; most of sites were old, large, with scarce environmental controls, received hazardous wastes and caused a primary exposure pathway through inhalation of volatile organic compounds (VOC). The authors of the review concluded that there was little consistency in the results and that, on balance, there were more studies reporting no associations than those reporting positive associations. The most frequent adverse health outcomes associated to residence near landfills were birth defects and low birth weight while the number of studies reporting increased risks of various self-reported symptoms could have resulted more from subjective anxiety associated with the residence near a landfill than from direct effects of chemical releases.

Health hazards due to different methods of waste management were reviewed by Rushton (2003). The author concluded that, while the occurrence of congenital malformations and the increased incidence in low birth weight near landfill sites has been reported several times, there is a general lack of consistency in the results for cancer incidence and mortality studies. Limits of the studies were discussed and it was stressed that, in many of these studies, data on potential confounders and exposure information is poor and latency of diseases and migration of populations are often ignored.

In 2004 the United Kingdom Department for Environment Food and Rural Affairs (DEFRA) commissioned Enviros Ltd and University of Birmingham a review of the health and environmental impacts of waste management (Enviros Consulting Ltd et al., 2004). The section dedicated to landfills health effects principally relied on the study carried out by Redfearn and Roberts and did not bring any further element of analysis.

As supplementary material for the book “Child Health and the Environment” (Wigle, 2003) a review on epidemiological evidence on the health effects of hazardous waste sites, focused on children and updated to March 2004, has been published on the web (Wigle, 2004). Summary results, synthetically reported in Table 2, described the level of epidemiological evidence as “inadequate”¹ or “limited”².

¹ For “inadequate” it is meant: relationship for which epidemiological studies were limited in number and quality (for example, small studies, ecologic studies, limited control of potential confounders), had inconsistent results, or showed little or no evidence of exposure-risk relationships (Wigle, 2003).

² “Limited” evidence includes relationships for which several epidemiological studies, including at least one case-control or cohort study, showed fairly consistent associations and evidence of exposure-risk relationship after control for potential confounders (Wigle, 2003).

Table 2. Hazardous waste disposal sites: summary of epidemiological evidence

HEALTH EFFECT	LEVEL OF EVIDENCE	HEALTH EFFECT	LEVEL OF EVIDENCE
Early fetal deaths (spontaneous abortion)	Inadequate	Oro-facial birth defects	Inadequate
Late fetal deaths (stillbirths)	Inadequate	Musculoskeletal birth defects	Inadequate
Intrauterine growth retardation	Inadequate	Genitourinary birth defects	Limited
Small for gestational age	Inadequate	Gastrointestinal birth defects	Inadequate
Birth weight adjusted for gestation length	Inadequate	Chromosomal abnormalities (structural)	Inadequate
Term birth weight	Inadequate	All childhood cancers	Inadequate
Low birth weight (not adjusted for gestation length)	Limited	Leukaemia	Inadequate
Preterm birth, gestation length	Inadequate	Lymphoma	Inadequate
Total birth defects	Limited	Reproductive system development	Inadequate
Central nervous system birth defects	Limited	Thyroid function	Inadequate
Cardiovascular birth defects	Limited	Kidney function	Inadequate

Source: adapted from Wigle, 2004

Linzalone and Bianchi (2005) updated current reviews on epidemiological studies on health effects of waste landfill sites discussing the results of seven ecological studies, two health monitoring investigations and two environmental investigations performed between 2000 and 2004. No consistent results in studies on cancer incidence, mortality and congenital malformations were reported. Increases in low birth weight and different types of symptoms were consistently found. The availability of environmental data and individual measurements of exposure was very poor in most of the studies. It was recommended by the authors to develop a new generation of epidemiological studies based on the use of biomarkers and environmental monitoring data.

The last available review was published in 2006 (Perez, Frank & Zimmerman) and dealt with the health effects of municipal solid waste (MSW) on individuals involved in their collection, transport, transfer and management. The authors reported that only few studies, characterized by poor exposure assessment, evaluated the relationship between organic dust exposure and health effects in occupational groups: associations between job involving the handling of MSW and various respiratory, dermatologic and gastrointestinal health effects were suggested.

Since Vrijheid's review (Vrijheid, 2000) two multi-site studies have been published. In the first study about cancer risks in population living near more than 9000 landfills in Great Britain, no excess risks of cancers (bladder, brain, hepatobiliary cancer or leukaemia) were reported (Jarup et al., 2002). In the second study, an Italian ecological correlation analysis carried out in the Campania region, excess in mortality cancer rates for stomach, kidney, liver, lung, pleura and bladder were found in an area characterized by illegal dumping and incineration of toxic and urban wastes (Comba et al., 2006). The results of a second phase of the study, not yet published, confirmed excess risks for some specific cancer causes (liver, stomach).

Concurrently with the workshop three multi-site studies were published, two of them dealing with United States hazardous sites. In the first one (Kuehn et al., 2007) a series of significant risks for congenital malformations, decreasing with distance from the sites, have been found; in the second one (Mueller et al., 2007) fetal deaths for women residing near the sites were not associated with the distance but an association was observed among women residing less than one mile from pesticide-containing sites. The third study (Jarup et al., 2007) analyzed the risk of giving birth to a child with Down syndrome, associated with residence near 6 289 landfill sites (processing special, non-special and unknown waste type) in England and Wales. Postcodes within the two kilometres zone were classified as exposed and people living beyond two kilometres comprised the reference population. A Bayesian regression model was applied to calculate relative risks and adjustments were made for major confounders. No excess risks of Down syndrome related to landfill sites were found and adjustment for socioeconomic status did not influence the estimates. No differences in risk between hazardous waste sites and other landfill sites were found.

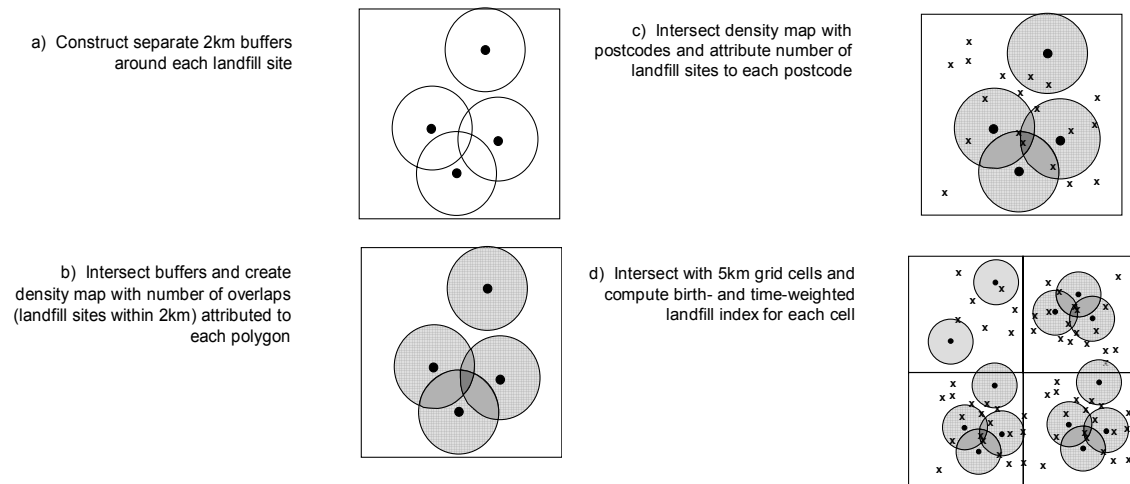
2.3 Critical case studies

Three case studies were presented in the framework of the discussion on health effects of landfills.

In the first case study, carried out at the largest chemical landfill in Kaergaard, Denmark (see Annex A13), the contamination of a beach due to the release of 280 000 tons of pharmaceutical and other chemical waste in pits located in the beach dunes was investigated. Chemical composition was analyzed, dispersion of the chemicals and their flux to the ocean was monitored through numerous measurement campaigns performed periodically. Concentration of chemical substances is constantly measured outdoor and in the air one metre below the surface. Since population living in the area is scarce and the beach is remotely located, access to the site was not forbidden to the public. In order to prevent health consequences, the erection of warning signs was considered a sufficient measure, considering the expected high level of compliance of local people to these warnings.

In the second case study, carried out in Great Britain, the health status of 80% of its population, living within 2 kilometres from landfill, was assessed; this is the largest study to date to report on the possible association between residence near landfill and health outcomes. More than 9000 sites (81% non-special waste and 19% special waste sites) were included in the study. Slight excess risks for low birth weight and congenital anomalies were found but no causal mechanism was identified to explain these findings, and alternative explanations could include data artefacts and residual confounding (Elliott et al., 2001a). No increased risk was found for cancer (Jarup et al., 2002). Analyses were repeated only for congenital malformations and a measure of landfill intensity, as described in Figure 1, to elucidate if risks were ubiquitous or driven by localized excesses.

Figure 1. Measure of landfill intensity in Great Britain



Source: David Briggs and Kees de Hoogh, presentation made during the workshop

Small excess risks were found in connection with intensity of special waste landfill sites for hypospadias/epispadias and neural tube defects; no excess risk was on the contrary associated with intensity of non-special sites (Elliott et al., 2006). The authors concluded that the evidence suggested a slight relative risk from landfill sites that, given the large number of people potentially exposed, could represent a significant public health issue. They also noticed that socioeconomic gradients in subpopulations defined by exposure level leave open the possibility of residual confounding.

In the third case study, several epidemiological studies carried out around the municipal waste landfill in Turin, Italy (see Annex A14), were described. A first microgeographical study, carried out without considering individual data by WHO and the Regional Agency for Environmental Protection of Piedmont Region (ARPA Piedmont) (Mitis et al., 2004), found out that for increasing distances from the landfill site declining trends of risk for some causes of death were identified. Concerns for the results caused in-depth studies (Demaria et al., 2004, Ivaldi, Demaria & Cadum, 2003) that followed up individuals for more years, analyzed hospital admissions and congenital malformations and focused on the occupational and economic context. Developing distance regression models from a point source it was found that several excess risks were due to factors as education, housing type, occupation and socioeconomic deprivation. Respiratory hospital admissions in women became statistically significant in excess only after all those covariates were included in the model. After these studies the construction of an incinerator, that should have been located in the same area, was cancelled and realized elsewhere, as an act of environmental justice.

3. Incinerators

Solid urban waste incineration started at the end of nineteenth century. The first incinerator, called “Destructor”, was built in 1876 in Manchester, by Alfred Fryer (Maxwell, 1967) and was originally introduced for reasons of hygiene and volume/weight reduction. In 1893 an incinerator producing steam existed in Hamburg and between 1903 and 1905 there were two plants for district heating and cogeneration in the United States. But only at the end of the 1960s, to decrease the pollutant emissions in the atmosphere, were incinerators more frequently equipped with energy recovery systems (Beltz, 1979).

Nowadays, incineration represents only a part of a complex waste management system that should include reduction of production, differentiated collection and re-use of waste, recovery (of materials and energy) and final disposal. The goal of current waste incineration technology is to treat waste so as to reduce its volume and hazard, to capture, concentrate and destroy potentially harmful substances and to recover energy from combustion.

3.1 Emissions and exposure

Waste fuel for incinerators are crude MSW, residual from differentiated waste collection and treated MSW or refuse derived fuel. Inorganic emissions include water (vapour), carbon oxide (CO), carbon dioxide (CO₂), sulphur oxides (SO_x), nitrogen oxides (NO_x), and products of incomplete combustion such as silicates, inorganic ash, soot, metal elements and their oxides and salts (for example, mercury and other metals with high vapour pressure). Organic emissions include VOC, hydrocarbons (HC), dioxins (polychlorinated dibenzo-p-dioxin (PCDDs) and polychlorinated dibenzofuran (PCDFs)), polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs). Particles (particulate matter with an aerodynamic diameter smaller than 10 (PM₁₀), 5 (PM₅), 2.5 (PM_{2.5}) microns and ultrafine particles) are emitted too. Further emissions, not related to the stack, can be summarized by ash, bottom ash, fly ash, noise, odour, pests, transport-related emissions, dusts and spores.

Incineration plants emission limits given by European Directives are described in Table 3.

Table 3. Incineration plants emission limits: the European Directive

POLLUTANT	LIMIT (mg/Nm ³ s)	POLLUTANT	LIMIT (mg/Nm ³ s)
Total dust	10–30	TOC ¹	10–20
HCl ²	10–60	Cd ³ , Tl ⁴ , Hg ⁵	0.05*
HF ⁶	1–4	Sb ⁷ , As ⁸ , Pb ⁹ , Cr ¹⁰ , Co ¹¹ , Cu ¹² , Mn ¹³ , Ni ¹⁴ , V ¹⁵	0.5
SO ₂	50–200	PAH	-
NO ₂ ¹⁶	200–400	PCDD + PCDF (ng/Nm ³)	0.1**
CO	50–100		

Note: the double limit value is: daily average and maximum (hourly or 30 minutes average).

¹Total organic compound; ²Hydrogen chloride; ³Cadmium; ⁴Thallium; ⁵Mercury; ⁶Hafnium; ⁷Antimony; ⁸Arsenic; ⁹Lead; ¹⁰Chromium; ¹¹Cobalt; ¹²Copper; ¹³Manganese; ¹⁴Nickel; ¹⁵Vanadium; ¹⁶Nitrogen dioxide.

*Limit for (Cd+Tl) and Hg separated.

**Calculated using the concept of toxic equivalence factors referred to 2,3,7,8 T₄CDD.

The stack height is never less than 70 metres but in modern large plants can be higher (up to 120 metres). The stack effective height (geometric plus enthalpic thrust), local atmospheric conditions and topography situation determine the dilution of emissions.

Incinerators have been operating in many European countries since 1960s–70s and their technology has evolved over time, with general reduction of emissions affecting neighbourhood communities. New generation incinerators built with the “best available technologies” (BAT) are characterized by a flue–gas multistage cleaning treatment and guarantee emissions within the limits specified by the European Directive. These plants do emit pollutants into the environment but it is unlikely that they would make a significant contribution to the overall background level of air pollution in a particular area if properly run and maintained and if adequate waste is processed.

Populations living near incinerators are potentially exposed to chemicals by way of inhalation of contaminated air, consumption of contaminated foods, water or dermal contact with contaminated soil (Franchini et al., 2004). However, the presence of hazardous agents in the vicinity of an incinerator cannot easily be translated into useful exposure information. In fact, it is difficult to assess if human exposure has occurred at all, let alone quantify it, although for few substances it is possible to use biomonitoring to identify exposure. Surrogate measures of exposure such as the distance of residence from the plant are thus often used in epidemiological investigations; these proxies have many limitations, for example they can be inadequate if other factors (stack height, prevailing wind direction, fall out area) are not considered.

The location of the plant is a critical factor. Most incinerators are located in areas where many other sources of exposure are present: caution is therefore needed in the interpretation of occurrence of high disease risks, which may be due to risk factors other than the incinerator emissions.

Occupational exposure is of higher intensity and duration but results of these studies are difficult to extrapolate to the general population because of a set of confounders (sex, age, lifestyle) and of the healthy worker effect.

3.2 Scientific evidence

Compared to landfills, a limited number of epidemiological reviews were carried out to assess the health effects of incinerators on the population occupationally exposed or living in the surroundings.

In four out of six studies reviewed by Rushton (2003) excesses were reported for specific cancer causes (cancer of the digestive system, of the liver, kidneys, pancreas and NHL); some excesses for skin, stomach and respiratory cancer were reported in occupational studies together with a plausible strong association for low birth weight. Exposure data were found to be inadequate for drawing reliable conclusions on these associations.

Franchini et al. (2004) reviewed articles published between 1987 and 2003. A total of 45 articles were selected: 32 concerning health effects on population living near the plants, 11 on occupational exposure and 2 on both environmental and occupational exposures. In most of the studies exposure was poorly characterized because of the lack of information on emission, nature of waste feed and off-site migration routes from the incineration sites. The majority of the studies concerned first generation incinerators, characterized by limited abatement technology and low combustion temperatures, resulting in high emissions. The emissions of more modern incinerators investigated in studies included in the review are more limited and of different kind. For this reason the results of all the studies cannot be easily compared, and consistency across studies is not expected. However suggestive evidence was provided by two thirds of those studies focusing on cancer. In these studies, significant positive results were found for some specific cancers (NHL, STS, lung cancer and childhood cancer); results on cancer of larynx and liver were not consistent. Results on non-cancer end-points, such as chronic or acute respiratory

effects in children or adults, were inconclusive. Occupational studies provided some evidence on lung cancer, esophageal cancer, blood PCDD/F level; an increased risk of producing urinary mutagens in exposed workers was reported. Biomonitoring studies did not provide conclusive evidence: in some studies exposure to PCB and heavy metals were associated with reduction of thyroid hormones. The authors suggested the need of carrying out further research based on the development of specific biomarkers and through the implementation of systematic environmental measurements to better characterize the exposure.

In the review carried out by DEFRA (Enviros Consulting Ltd et al., 2004), already mentioned above, it is concluded that there is no convincing evidence of association between incinerators and cancer while there is a little evidence on respiratory problems. It is however stressed that in most cases the incinerator emissions make a small contribution to local levels of air pollution.

A recent publication in Italian reviewed papers included in the Medline literature database, published between 2003 and March 2006, identifying 32 publications (Bianchi, Franchini & Linzalone, 2006). Most of the studies assessed the individual exposure using biomarkers, while “traditional” epidemiological studies, based on surrogate exposure metrics, decreased in number. Occupational studies consistently identified job categories at higher risk for exposure to fly ash, particles, metals, organic compounds, dioxins and positive association with length of employment. Results from studies on the general population were less consistent: some associations for NHL and STS were reported but the number of studies was limited and exposure was poorly characterized.

Other European studies were analyzed in a recent review (Linzalone & Bianchi, 2007). The authors focused on fine and ultrafine emissions coming from incinerators and stressed the necessity to adopt a precautionary approach, because of the limited number of studies and the weakness of the present knowledge on the health effects. They described two French studies (Institut de Veille Sanitaire, 2006a, b), dealing with the relation between dioxin levels in blood of population living near incinerators and cancer incidence, that did not give consistent results and an Italian metanalysis (Bianchi & Minichilli, 2006) that found an excess of male mortality for NHL in the population of 25 small municipalities with municipal solid waste incinerator (MSWI). A significant association between modelled dioxin exposure and sarcoma risk was detected in the Province of Venice (Zambon et al., 2007).

More recent Portuguese and French studies are described in the case studies section (see paragraph 3.3).

3.3 Critical case studies

Three case studies, described in detail in section 6, were presented to facilitate the general discussion on health effects of incinerators.

The first case study, carried out in Barcelona, Spain (see Annex A15), considered the MSWI of Mataró (Gonzales et al., 2000, Gonzales et al., 2001). Nineteen per cent of waste produced in Catalunya is burned in this plant, with a satisfactory energy recovery, emissions largely within legal limits, but with an insufficient capacity. A biomonitoring study was initiated because of local community concerns over the effects of dioxins. Study subjects were recruited among volunteers and classified as potentially exposed or unexposed according to the distance of their residence from the plant. A control group from another city without an incinerator was also included in the study. Dioxin, furan and PCB levels in the blood were repeatedly measured and a questionnaire on sociodemographic factors, occupation, dietary habits (consumption of locally grown foods), reproductive history and respiratory symptoms was distributed. Average dioxin levels were found, unexpectedly, to grow in all study subjects over time (40% in 4 years); high

concentrations were not limited to people living near the incinerator, affected both sexes and were independent of age. The authors concluded that the increases in dioxin levels could not be attributed to incinerator emissions but probably reflected an increase in exposure to and intake of dioxins from food or other unknown sources.

In the second case study, several epidemiological studies (Floret et al., 2003, Floret et al., 2004, Floret et al., 2006, Viel et al., 2000) carried out around the municipal waste incinerator in Besançon, France (see Annex A16), were described using a sequential approach starting from crude investigations, gradually refining towards specific, aimed studies, following what the author referred to as a “funnel” type approach. Such sequence included, in order, a macro-spatial step, a micro-spatial step, the validation of a diffusion model, the dioxin measurements in locally produced food and a case-control study with dioxin blood levels. With the macro-spatial step (Viel et al., 2000) clusters of NHL and STS were first found near the incinerator. In the micro-spatial step (Floret et al., 2003, Floret et al., 2004) analyses used a case-control study design applied at block level (a block has 161 inhabitants on average), including measures of socioeconomic status; increased risk was found for highest dioxin levels, derived from a validated atmospheric diffusion model (Floret et al., 2006). The next study in the sequence, based on measured levels of dioxin and heavy metal in locally produced eggs and vegetables, also evaluated in a case-control design in connection with dioxin and pesticide blood levels, is underway; results are due in one year.

In the third case study (see Annex A17), carried out on the population living near two Portuguese incinerators (one in Lisbon and one in Madeira Island), a large monitoring project was described (Reis et al., 2007a, Reis et al., 2004a, Reis et al., 2007b, Reis et al., 2007c, Reis et al., 2007d, Reis et al., 2007e, Reis et al., 2004b). These HBM studies set out to investigate the local exposure to the most critical incineration-related pollutants: dioxin and heavy metals levels were measured in the blood of the general population, lead and mercury levels in the blood and pubic hair of mother-child pairs and in children under six years of age, and dioxins levels in the milk of breastfeeding women. Analyses and surveys of possible adverse health effects were conducted on asthma prevalence, cancer mortality and incidence, mental health status, self-perception of health status and frequency of reproductive disorders such as low birth weight, preterm delivery, spontaneous abortion, foetal malformations and infant and perinatal mortality. The biomonitoring campaigns were based on a series of cross-sectional studies, repeated every two years and, in order to control the confounding, on questionnaires on anthropometric and sociodemographic factors, lifestyle and behavioural variables. Epidemiological studies were carried out through periodic analyses of health registries (cancer mortality, infant and perinatal mortality, low-birth-weight and foetal malformations data) and on self-administered questionnaires dealing with asthma and tobacco consumption, mental health and self-perception of health status. Results from the surveillance programs showed no statistically significant differences for blood level of dioxins and for health outcomes in exposed and not exposed population (defined according to the distance from the plants) suggesting the presence of effective source control measures and abatement technology in both plants. Results from exposure determination showed a significant declining trend for almost all the pollutants (dioxins and metals) under study.

4. Overall assessment of health impact of waste

An overall assessment of the available literature on waste management and health cannot be done without addressing landfills and incinerators separately, given the differences between the two in terms of the toxic agents involved, the routes of exposure, the health effects and impacts, the degree of evidence and the relevant policy responses.

4.1 Landfills

All studies that have been reviewed in this field have important limitations, as follows.

- Exposure misclassification: none of the studies have had access to personal exposure data. There is limited consistency in terms of the distances used as a proxy for exposure. Distances from the site(s) that have been used vary from 240 metres up to 4.8 kilometres; distance is unlikely to be an accurate measure of exposure to atmospheric pollutants and even less knowledge exists about exposure via other pathways, for example water and land.
- Case ascertainment: several studies, for example on resident population, may have resulted in loss of informative cases; also national registries and databases may be incomplete.
- Confounding factors and bias: there is no consistency in the control for confounding factors. While deprivation has not been shown to be an important factor in some European studies, it was found to play a role in some United Kingdom and United States studies. A single snapshot of area deprivation may not fully control for this confounder, though no alternative indicators are routinely available.
- Multiple testing: several studies have conducted multiple tests increasing the chance of a Type I error (false positive).
- Site contents: little attempt to characterize site contents let alone emissions or exposures. Some studies have been restricted to hazardous solid waste but this itself is a broad category and provides little intelligence on the actual site content. Multi-site studies aggregate populations but do not account for the enormously varied contents of different sites and may dilute any real associations with specific pollutants.
- Populations and general disease categories: single-site studies will not have adequate populations to give sufficient power to demonstrate a small but important effect. Multi-site studies aggregate data but those sites with larger populations will exert the greatest influence on the analysis. Specific congenital anomalies may have different aetiologies that cannot be identified in pooled studies.

Despite these limitations, the scientific literature on the health effects of landfills provides some indication of the association between residing near a landfill site and adverse health effects. The evidence, somewhat stronger for reproductive outcomes than for cancer, is not sufficient to establish the causality of the association. However a public health response is warranted because a small but significant excess risk of several reproductive adverse outcomes, together with the large proportion of population potentially exposed to landfill, and the level of available evidence suggest that the potential health implications cannot be dismissed.

Given the difficulty of using residence location as a proxy for exposure and the long latency, fewer studies concern cancer. In combination with poor exposure classification, this can lead to very low power of the study to find a real risk.

4.2 Incinerators

With regard to incinerators most of the points raised for landfills are valid. It has to be stressed that most of time confounding makes studies hard to do and even harder to interpret and, as in landfill studies, increases in relative risk are difficult to detect because are generally caused by long-term low-level exposures. Studies pointing to an increase in STS and NHL support a possible etiologic role of 2,3,7,8 T₄CDD. The evidence is inadequate to draw conclusions that are valuable for guiding current policy choices on incineration: relatively few good quality studies exist and they refer mostly to old generation incineration plants. In addition in some studies in which risk excesses were found, alternative interpretations, for example involving exposures from sources other than the incinerators were put forward.

It is important to point out that stack emissions from modern plants are much reduced compared to old generation plants. The few studies carried out on new generation incinerators are difficult to compare with the previous ones, because of these differences in technology between the plants. The adoption of the BAT, enforced by the EU, results in the fact that the occurrence of measurable health effects on populations resident in close proximity of new generation incinerators is becoming less likely. However their overall impact on the general environment and on human health through indirect mechanisms of action, has not been evaluated yet. In particular waste incineration, currently on the increase in many countries, may be a non-negligible contributor of greenhouse gases and persistent pollutants on a global scale.

5. Discussion points: knowledge gaps, uncertainty, research priorities, action and policy options

With regard to the health effects hypothetically caused by both incinerator and landfills, further insights are unlikely to be gained from distance-based studies, given their substantial limitations. If progress is to be made, it seems necessary to consider exposure pathways and biomarkers of exposure and effect, and compare waste-related exposures with those due to other sources of pollution. A first step towards “new generation” epidemiological studies that can better characterize exposure through the use of biomarkers has already been taken, especially in the case of incinerators.

The public concern about congenital malformations has the potential to deflect research efforts from investigating other health effects, for example renal and liver disease, that may have comparable public health relevance, but is less prominent in the policy debate. This may in turn create a tendency to investigate what has been investigated before – for example driven by better chances of publishing results in the peer-reviewed literature – or to study health outcomes for which information is most readily available. For these reasons an underestimation of adverse health effects is possible.

The evidence of adverse health effects related to landfills and incinerators should add to other environmental concerns in moving waste management strategy to waste minimization and recycling, as prescribed by EU Directives. In the transitional period, countries should move toward zero tolerance of poor waste management practice, supported by resources for regulation and enforcement, and invest in technology for lowering emissions to air and water.

Decision-making concerning location of waste facilities should be transparent, fair and rational. In some cases (see Campania region), as waste management moves from a poorly managed or even illegal regime to legal and properly managed operations, long transitional periods can be expected, as well as considerable uncertainties on the implications in terms of exposures to environment and health.

In fact, the main uncertainties encountered in health risk assessments of waste management practices can be classified as specific to the issue of waste management and generic to the practice of health risk assessment.

Sources of uncertainty specific to the waste management issue include the data on waste generation and waste management practices, data on emissions, exposure characterization and consequently exposure-response characterization. For example, the assessment of the health risks posed by composting, an important component of waste recycling, is expected to be an area of high uncertainty. There is often very little empirical information available on the quantity and nature of the emissions from composting operations and very little empirical information on the health effects.

In addition to the inconsistent quality of the data describing waste composition and volumes, there is a great lack of data concerning wastes that are illegally disposed of. Anecdotal evidence indicates that the illegal disposal of waste is a frequent practice in some areas in Europe. Because a variety of illegal disposal practices exist (burning, dumping, etc...) and because it is very difficult to estimate the amounts of waste that are disposed of illegally, determining emissions and exposure levels is virtually impossible.

While emissions from waste management facilities are often estimated using emission factors reported in the literature, these emission factors are subject to some uncertainty. Amongst other

things, there is a paucity of data describing the emissions of some waste management options such as composting and gasification. Furthermore, the emissions factors reported in the literature are often derived from facilities under normal conditions of operation, making it difficult to account for improper operations or accidental releases. Also, emission rates often depend on the effectiveness of the technology in use, causing large variations between old and new facilities. Thus, transposing emission factors determined on the basis of a certain vintage of technology to other locations using newer or older technology leads to uncertainty.

A common approach to exposure characterization is to use distance from the site as a proxy for exposure. This leads to uncertainty in both the determination of the exposure–response function, as well as in the exposure assessment. In some cases, the small number of people living in the vicinity of waste management facilities makes it difficult to achieve sufficient statistical power in epidemiological studies. Conversely, in the case of landfill sites, studies such as that of Briggs et al. (presented at workshop) indicate that urban areas are typically marked by large concentrations of abandoned landfill sites, thereby making it difficult to achieve a good level of contrast in exposure.

While in the case of some pollutants like PM₁₀ the exposure–response functions are well defined, for other pollutants such as sulphates, nitrates and organic compounds, these functions are characterized by high levels of uncertainty.

Given the state of the available knowledge, the surrounding uncertainties, the potential for health damage and the widespread concern over the issue, several actions appear to be appropriate.

The decision to adopt epidemiological surveillance programs in areas characterized by the occurrence of landfills or incinerators should be taken on the basis of a feasibility analysis aimed at avoiding the execution of non-informative studies. In the cases in which epidemiological surveillance appears to be appropriate, suitable protocols should be adopted, especially evaluation research after major interventions.

Regardless of the final decision on the appropriateness of a local epidemiological study, actions aimed at meeting population's concerns should be adopted, namely:

- provide information on technological standards and process characteristics and environmental mitigation strategies; resources should also be concentrated on establishing the real level of risk associated with sites, including improved understanding of exposure pathways, before considering site specific epidemiological research;
- provide information on environmental monitoring;
- develop monitoring programs (if applicable); and
- enhance communication and participatory activities in order to promote community autonomy.

In this framework, a specific role for scientific research is foreseen. Epidemiological, toxicological, exposure–assessment studies have not yet sufficiently and adequately incorporated HBM data for evaluating individual, community and population health risks. This is needed to design and perform biomonitoring–based studies and surveillance, especially to investigate health effects, due to low-level environmental chemicals exposure, that may be felt over a long time period. For these reasons priority needs for research include:

- in the epidemiological field, HBM studies are needed to improve understanding of the relationships between HBM data and health effects; attention should be given to number of biosamples collected/stored (also considering the future research opportunities);

- in the toxicological field it is necessary to carry out experimental studies based on HBM to assess biomonitoring dose–response relationships;
- pharmacokinetic models should be used to assess the influence of factors as metabolism and sampling time, essential to understand biomonitoring response; and
- a detailed analysis of the environmental matrices state to evaluate chemical exposure pathways is needed to assess the exposure for specific subsets of the population.

The integration of research monitoring, public health action and communication process may be achieved through integrated and participatory approaches, such as HIA. In fact, evaluation of the potential impacts of waste, both on environment and health has been significantly aided by the use of impact assessment methodology. EIA generally focuses on negative environment and health impacts at a project level. HIA can be done at a policy, program or project level to judge the potential effects of a proposal as well as the distribution of those effects. Understanding and managing the potential or likely health impacts of waste management is likely to be best addressed through either HIA or SEA.

In this frame, the following issues should be the object of particular attention.

- The health impact of waste management procedures cannot yet be properly evaluated, because of the afore-mentioned limitations of the current state of knowledge. However, absence of evidence is not evidence of absence. Hence, available information of localized environmental contamination and, to some extent, of increased occurrence of adverse health effects in the vicinity of landfills and incinerators should inspire a precautionary approach with respect both to the installment of new facilities and the mitigation of exposure to emissions and leachates of existing sites.
- Priority–setting for environmental remediation should be based on hazard detection, estimation of the size of the exposed population (including vulnerable groups) and appreciation of inequity in the distribution of exposure among population subgroups, taking into account the possibility of higher exposure of socioeconomically deprived groups. The latter aspect should be present to decision-makers even if the absolute number of exposed subjects might appear to be small, thus resulting in a moderate number of attributable cases.
- The questions of distribution of impacts and benefits, as well as the perceived risks, should be taken into consideration when comparing the health implications of waste against known risk factors with established higher effects (for example, traffic–related air pollution or passive smoking).
- This process should be made accessible to administrators and concerned communities in a truthful and understandable way, taking into account local concerns, promoting participatory activities and supporting autonomous decision processes.

Annex A. Extended abstracts

A1. An overview of health effects of landfill sites

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The exposure pathways and types of exposure associated with landfills are complex:

- involving air, water, soil and food;
- involving inhalation, ingestion and skin contact;
- involving a wide range of different chemicals, some of them known carcinogens, allergens, or teratogens if exposures are sufficiently high and present in chemical mixtures with unknown synergistic effects; and
- relating to landfilling practice, often many years before the time of exposure which may have been poorly documented or illegal.

In addition, there is transport-related pollution, noise, litter, smell, impact on house prices, poorer visual amenity and, though less studied, potential for infectious exposures.

It is not surprising in the face of this complexity, that most health studies to date have studied those living near landfill sites as a “proxy” for complex exposure, but also in order to address the question of how much health cost those living near sites are experiencing, for the benefit of the whole community, essentially an environmental justice question. It should be remembered, however, that landfill sites can affect the health of the wider community through pollution of groundwater used in drinking-water distribution, or contamination of agricultural produce grown near the site, or through the global warming effects of methane emissions.

A number of landfill site characteristics affect the likelihood that a landfill leads to potentially harmful population exposure. These include site engineering, hydrogeology and topography, the type and quantities of waste deposited, the mixing of contents and, last but not least, the management practices that avoid emissions to water and air, and major accidents. Given the heterogeneity of sites, we do not expect consistency between studies of health effects around different sites.

A distinction can be made between single-site and multiple-site epidemiological studies. Single-site studies cannot easily demonstrate that the excess ill health of the population is due to the landfill site, rather than some other characteristic of the local population. Single-site studies which start with the observation of a cluster of adverse health outcomes are particularly difficult to interpret, since unusual aggregations of adverse health outcomes do occur by chance, and there may be no causal relationship with any local exposure. However, some single-site studies have usefully emphasized the temporal dimension, showing for example that excess low birth weight was specifically related to the time periods of highest a priori estimated exposure (Berry & Bove, 1997, Kharrazi et al., 1997), assuming of course that this estimation was correctly made, or conversely that an excess of health effects was present both before and after the site opening (Fielder et al., 2000). Others have used olfactory zones assuming that air emissions are correlated to odour, or contrast populations up and downwind (Vrijheid, 2000). Where there was a specific incident of landfill-related pollution of drinking-water, it has been possible to document also which residents drank more of this water (Vrijheid, 2000). There is a limit, however, to how much such categorisation of the local population can be pursued, because of the overriding problem of achieving a great enough population size to detect an increased risk of plausible size, especially when studying rare health outcomes.

Multi-site studies are statistically more powerful to detect lower levels of excess risk and are essential for rarer but severe adverse health outcomes such as congenital anomalies. However, they generally are able to conduct less extensive exposure assessment and/or description of exposure. Misclassification of exposure tends to dilute estimates of relative risk. Causal interpretation can be greatly strengthened by showing a dose-response effect either within sites or between sites of differing hazard potential (Dolk et al., 1998, Geschwind et al., 1992, Vrijheid et al., 2002b), but this again presumes an accurate relative assessment of the overall hazard of complex exposure which can be difficult to achieve. Multi-site studies should control for important confounding. Since low socioeconomic status is associated with a range of adverse health outcomes, and can also be associated with residence near landfill sites, control for socioeconomic confounding is necessary. Other than socioeconomic status, it is not easy to postulate a risk factor that would be systematically associated with proximity to multiple landfill sites. Two such

factors may be other industrial sources of exposure in the vicinity, or heavy traffic (including landfill-related traffic). Future studies should attempt to “map” major pollution sources in order to address this issue. In summary, excess risk shown in a multi-site study with suitable control for socioeconomic confounding can be interpreted as “possibly due to other factors, but, if real, likely to be higher than estimated”.

There are now a number of well conducted studies showing excess risk of congenital anomalies and low birth weight, and others that are less convincing or fail to find any effect. The resulting body of evidence can be called “suggestive” rather than “inconclusive”, while acknowledging the interpretational problems discussed above. What they suggest is that some landfill sites pose a risk to the reproductive health of the nearby population, but they cannot as yet give a reliable indication as to which sites pose such a risk. Nevertheless, it can be inferred from what is known of landfill that poor management and poor containment, particularly where there is handling of quantities of hazardous waste, are likely to contribute to the level of hazard, and a rigorous system of site inspection with openly accessible reports should be implemented. Positive biomonitoring studies, currently lacking, showing high levels of real exposure near some sites and controlling for confounding, would greatly strengthen the causal interpretation of the adverse pregnancy outcome studies. Negative biomonitoring studies, however, may simply have measured the “wrong chemical” or ignored chemical combinations, and biomonitoring will not completely resolve the problem of causal inference.

Fewer studies relate to cancer, because of the extreme difficulty of using residence location as a proxy for exposure when individuals migrate over the long latency periods until cancer develops. In combination with poor exposure classification, this can lead to very low power of the study to find a real risk. Realistically, cancer studies may need to either concentrate on risks in industrialized areas (rather than specifically landfill), or use case-control techniques with interviews as to residential histories, a technique that has been used in relation to renal disease and landfill (Hall et al., 1996), in combination with more detailed measurements of exposure and confounding. Whether this is a worthwhile large-scale investment in research is not clear, as waste management strategies and decisions regarding particular landfill sites may not be critically dependent on the results, and more immediate benefits can be gained from exposure studies.

A further body of literature relates to “self reported symptoms” such as headache, fatigue, skin irritation, respiratory problems, and psychological problems. By necessity, these are single-site studies where the population can be interviewed and given medical examinations. Such studies suffer from other interpretational problems – incomplete and possibly biased response to the call for participants, as well as potential recall bias of symptoms by those worried about their residence near the site. However, the potential relationship between experience of symptoms and the presence of the landfill is complex – physical and mental health problems can arise from nuisance factors (smell, litter, vermin, noise), house price or saleability worries, and from the feeling of powerlessness in the face of environmental regulatory ineffectiveness. They may also arise from worry about the risk of other health problems as reviewed above, especially where this concerns children. Rather than argue over interpretation, the most effective way to address local concerns is to “clean up the mess” where it can be demonstrated that best practice is not being adhered to.

Problems with the review of evidence should be mentioned. The quality of a study cannot always be completely judged from the published paper, particularly the quality of the health outcome data. For example, incomplete ascertainment of congenital anomalies in the study population could lead to seriously biased results. Other problems include publishing bias toward positive findings particularly for single-site studies, and the scarcity of evidence about some health outcomes (such as liver and kidney disease or allergies), because studies tend to look at outcomes that have been looked at before or where medical information is most readily available. Multiple testing of many health outcomes (such as different congenital anomalies and different cancer types) inevitably result in some chance associations which must be confirmed in further studies, but there may be subsequent no comparable studies. Importantly also, there is a problem of extrapolation – what is relevant to one site or a set of sites, may not be relevant to others. Thus studies of contamination relating to old landfill practice may not be relevant to new sites with high quality engineering and management. The onus is on the waste management industry to ensure best practice always, and embrace environmental enforcement, in order to gain the trust of the population that a particular landfill can be “safe”.

Concerns about global warming have intensified to the extent that it is probably not the health concerns described above that will drive the waste management agenda toward waste minimisation and recycling. Nevertheless, the health evidence should add force to the agenda, and, moreover, add credence to the demands of residents that existing local landfills should engage in best practice, with zero tolerance of departures from best practice, and that siting decisions for new landfills needed in the transitional phase should be transparent, fair and rational and show the place of the landfill within a progressive national and regional waste management strategy.

A2. A systematic review of the evidence of an increased risk of adverse birth outcomes in populations living in the vicinity of landfill waste disposal sites

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A2.1 Introduction

A wide range of waste materials has been deposited in landfill disposal sites inevitably including known hazardous chemicals. Until recently, co-disposal of hazardous materials with municipal wastes was common in the United Kingdom and consequently there are potentially hundreds of sites that have received such wastes. Exposure to releases from landfills is feasible and it has been estimated that 80% of British people live within two kilometres of a landfill site. There have been several reports, reviews and commentaries suggesting health effects associated with living near such sites, particularly the risk of adverse birth outcomes including low-birth-weight babies or children born with congenital anomalies. This review examines studies of adverse birth outcomes in populations living near landfill sites using a systematic method.

A2.2 Methods

A broad search strategy was used to identify any literature, without language restriction, on the human health effects of exposure to landfill sites. Searches for published work were conducted using databases Medline (from 1966 to February 2006), Embase (from 1966 to February 2006), BIDS (1960–2006), Ingenta (1997–2006), Toxicology Abstracts, The British Library (Zetoc), Science Citation Index, Science Direct, Index to Theses and Scirus, and for ongoing studies, using the National Research Register. Reference lists of reviews were scanned for studies not retrieved in the searches. The search strategy was refined to focus on adverse birth outcomes as a key health outcome. All retrieved titles and available abstracts were assessed for relevancy to the review. Entire publications were obtained from those of possible value. The availability of unpublished studies was also investigated by contact with colleagues within the Health Protection Agency who have special interest in this area of study.

A2.3 Results

The review identified and assessed 29 papers examining the relationship between residential proximity to landfill sites and the risk of an adverse birth outcome. Twenty nine papers have been reviewed; 18 studies were multi-site and 11 single-site. Eighteen papers reported some significant association between an adverse reproductive outcome and residence near a landfill site. Twelve of these (Dodds & Seviour, 2001, Dolk et al., 1998, Elliott et al., 2001a, Fielder et al., 2001, Fielder et al., 2000, Geschwind et al., 1992, Goldman et al., 1985, Kuehl & Loffredo, 2003, Malik et al., 2004, Palmer, S. R. et al., 2005, Shaw et al., 1992, Vrijheid et al., 2002a) reported increased risk of congenital and seven (Baibergenova et al., 2003, Berry & Bove, 1997, Elliott et al., 2001a, Goldberg et al., 1995, Goldman et al., 1985, Kharrazi et al., 1997, Vianna & Polan, 1984) of low birth weight. Spontaneous abortion was reported in two (Fielder et al., 2001, Zejda et al., 2000) and mortality/still births in two (Kharrazi et al., 1997, Zejda et al., 2000). Prematurity was reported in one study (Dodds & Seviour, 2001) and gestational age in one (Kharrazi et al., 1997). Accordingly the discussion has concentrated on congenital anomalies and low birth weight. Nine of these studies (Baibergenova et al., 2003, Dolk et al., 1998, Elliott et al., 2001b, Geschwind et al., 1992, Kuehl & Loffredo, 2003, Malik et al., 2004, Palmer, S. R. et al., 2005, Shaw et al., 1992, Vrijheid et al., 2002a) were multi-site including four (Dolk et al., 1998, Elliott et al., 2001a, Palmer, S. R. et al., 2005, Vrijheid et al., 2002a) of the seven strongest papers (Dolk et al., 1998, Elliott et al., 2001a, Morgan, Vrijheid & Dolk, 2005, Morris et al., 2003, Palmer, S. R. et al., 2005, Vrijheid et al., 2002a, Vrijheid et al., 2002b) identified in this review. Two multi-centre, multi-site, case-control studies conducted under the European Collaborative Study of Residence near Hazardous Waste Landfill Sites and Risk of Congenital Malformations (EUROHAZCON) programme found similar moderate albeit significant associations between maternal residential proximity (within three km) to hazardous waste sites and both chromosomal (Vrijheid et al., 2002a) and non-chromosomal (Dolk et al., 1998) congenital anomalies, adjusted ORs of 1.41 (95% CI: 1.00–1.99) and 1.33 (95% CI: 1.11–1.59) respectively. A large national multi-site geographical comparison study (Palmer, S. R. et al., 2005) reported a risk ratio of 1.21 (95% CI: 1.04–1.41) after site opening and an after/before ratio of 1.39 (95% CI: 1.21–1.72). This study also reported a much a greater rate ratio for sites considered to represent the strongest exposure risk of 3.93 (95% CI: 1.43–16.95). A major national geographical comparison study (Elliott et al., 2001a) involving almost 10 000 landfill sites and over eight million births found a small albeit significantly increased risk of total and specific anomalies and low- and very low-birth-weight babies in populations within two km, RR of 1.01 (99% CI: 1.005–1.023), 1.05 (99% CI: 1.047–1.055), 1.04 (99% CI: 1.03–

1.05)) respectively. Two of the remaining five multi-site studies reporting an association are good quality geographical comparisons or case-control studies and reported similarly modest increased rate ratios or ORs for congenital anomalies of 1.12 (95% CI: 1.06–1.18) (Geschwind et al., 1992) and 1.5 (95% CI: 1.1–2.0) (Shaw et al., 1992).

A case-control study (Malik et al., 2004) reported similar ORs for congenital heart disease (1.2, 95% CI: 1.1–1.4) and a geographical comparison (Baibergenova et al., 2003) reported similar ORs for low birth weight (1.04, 95% CI: 1.02–1.07). A poor case control/geographical comparison study (Kuehl & Loffredo, 2003) found reported a very high OR of 13.4 (4.7–37.8) for a specific cardiovascular anomaly in areas containing four sites although there is real potential that an artificial cluster had been created.

Nine of the eleven single-site studies reported a significant association with at least one adverse birth outcome. These were five moderate or good geographical comparisons (Berry & Bove, 1997, Dodds & Seviour, 2001, Fielder et al., 2001, Fielder et al., 2000, Kharrazi et al., 1997), two retrospective follow-up studies (Goldman et al., 1985, Vianna & Polan, 1984), one cross-sectional (Zejda et al., 2000) and one case-control study (Goldberg et al., 1995). Two of the three geographical comparisons studies (Fielder et al., 2001, Fielder et al., 2000), examining congenital anomalies reported similar RRs of 1.9 (95 % CI: 1.3–2.85 and 1.23–2.95, pre and post site opening (Fielder et al., 2000)) and 1.9 (95 % CI: 1.3–2.9) post opening (Fielder et al., 2001). The third geographical comparison reported a lower RR of 1.25 (95 % CI: 1.04–1.51 (Dodds & Seviour, 2001).

Fielder, using odour complaints as a proxy for exposure, reported greater RRs in two studies (Fielder et al., 2001, Fielder et al., 2000) although found that an apparent cluster in the first study predated the landfill site (except for gastroschisis) and in the second study the authors could not exclude a possible artefact resulting from differences in reporting practices between hospitals.

Six studies specifically or principally examined low birth weight (the outcome most consistently reported in single-site studies (Vrijheid, 2000)). A EUROHAZCON case-control study (Morgan, Vrijheid & Dolk, 2005) reported no relationship and a large multi-site geographical comparison (Baibergenova et al., 2003) and a well conducted single-site case-control study (Goldberg et al., 1995) reported ORs of 1.04 (95% CI: 1.02–1.07) and 1.20 (95% CI: 1.04–1.39) respectively although a good quality single-site geographical comparison study (Berry & Bove, 1997) reported a much higher OR of 5.1 (95% CI: 2.1–12.3). Vianna and Polan (1984) found a significantly higher percentage of low-birth-weight babies in the potentially most contaminated area compared to the rest of the State ($p=0.009$) in a single-site retrospective follow-up study. A low powered retrospective follow-up study (Goldman et al., 1985) reported significantly more low-birth-weight babies in an area close to the Love Canal site with some indication of a dose-response effect. This study also reported significantly more birth defects in the vicinity of the site although this effect was largely seen in a specific subset of people, the homeowner community, and subjective parental recall was largely used to ascertain birth outcome.

However, ten of these “positive” studies also reported no relationship with some adverse birth outcome (Dodds & Seviour, 2001, Elliott et al., 2001a, Fielder et al., 2001, Fielder et al., 2000, Goldberg et al., 1995, Goldman et al., 1985, Kharrazi et al., 1997, Palmer, S. R. et al., 2005, Shaw et al., 1992, Vrijheid, 2000, Zejda et al., 2000) and eleven other studies reported no significant association with some adverse outcomes (Baker, D. B. et al., 1988, Boyle et al., 2004, Croen et al., 1997, Dummer, Dickinson & Parker, 2003, Hertzman et al., 1987, Marshall et al., 1997, Morgan, Vrijheid & Dolk, 2005, Morris et al., 2003, Orr et al., 2002, Sosniak, Kaye & Gomez, 1994, Vrijheid et al., 2002b). Eight of these examined congenital anomalies (Boyle et al., 2004, Croen et al., 1997, Dummer, Dickinson & Parker, 2003, Hertzman et al., 1987, Marshall et al., 1997, Morris et al., 2003, Orr et al., 2002, Vrijheid et al., 2002b) (five solely congenital anomalies) and three low birth weight (Baker, D. B. et al., 1988, Morgan, Vrijheid & Dolk, 2005, Sosniak, Kaye & Gomez, 1994). Two reported on mortality/stillbirth (Dummer, Dickinson & Parker, 2003, Sosniak, Kaye & Gomez, 1994), two on prematurity (Goldberg et al., 1995, Sosniak, Kaye & Gomez, 1994), one small for gestational age (Goldberg et al., 1995), one on spontaneous abortion and post partum difficulties (Baker, D. B. et al., 1988) and one on a variety of adverse outcomes (Hertzman et al., 1987). Nine were multi-site studies (Boyle et al., 2004, Croen et al., 1997, Dummer, Dickinson & Parker, 2003, Marshall et al., 1997, Morgan, Vrijheid & Dolk, 2005, Morris et al., 2003, Orr et al., 2002, Sosniak, Kaye & Gomez, 1994, Vrijheid et al., 2002b) and included three of the strongest papers identified; a multi-centre, multi-site case-control study (Vrijheid et al., 2002b) from the EUROHAZCON programme which also employed an expert panel approach to characterizing potential exposure, a EUROHAZCON multi-site study (Morgan, Vrijheid & Dolk, 2005) examining low birth weight as well as a large national study following the Elliott method (Morris et al., 2003). Four well controlled case-control studies (Croen et al., 1997, Marshall et al., 1997, Orr et al., 2002) found no association with either congenital anomalies or low birth weight (Sosniak, Kaye & Gomez, 1994). A well conducted geographical comparison study (Dummer, Dickinson & Parker, 2003) and a less well controlled case-control and cross-sectional study (Boyle et al., 2004) considered congenital anomalies. A small single-site geographical comparison study

(Baker, D. B. et al., 1988) also found no evidence of an association with a range of poorly defined adverse birth outcomes.

Four studies (Elliott et al., 2001a, Fielder et al., 2001, Fielder et al., 2000, Palmer, S. R. et al., 2005) included data collected before and after the opening of landfill sites. The Palmer and both Fielder studies suggested an excess risk of congenital anomalies after site opening (although in one case an excess existed before opening) while the large national study (Elliott et al., 2001a), reported an excess RR for some specific anomalies during the period before opening suggesting that factors other than landfill might be responsible.

A2.4 Discussion and recommendations

While an initial assessment of the direction of the literature suggests overall support for an association in that there have been more papers reporting an association between residential proximity to landfill sites and reproductive health effects, closer assessment challenges this view. While most studies reporting a link are good quality, over half report no association with some adverse birth outcome and most of the papers reporting no association are also well conducted. Indeed, the better quality studies tend to show either small or no effects. However there are methodological difficulties in studying this area and all such observational epidemiology is open to bias. In particular, all studies in this field are compromised by a lack of objective exposure data and there are general quality issues including case ascertainment, bias, confounding and multiple testing. This review considers that the evidence of an association with adverse birth outcomes is unconvincing. However, public concern is high and is likely to remain so as legislation facilitates public involvement in the regulation process. Resources should be concentrated on establishing the real level of risk associated with sites including improved understanding of exposure pathways before considering site specific epidemiological research, developing an environmental public health tracking system which combines environmental and health data to enable a more appropriate and routine assessment of the effects of environmental contamination (including landfill sites), and on developing an effective public information strategy to ensure that authoritative, impartial, timely and relevant information and advice are provided to the public.

A3. Overview of health effects – incinerators

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The use of incinerators is a common part of municipal waste treatment in most developed countries. However the practice remains controversial, and there is often public opposition to proposals for new incinerators. In this review we will briefly discuss some issues in risk assessment in general, then review the major emissions from waste incinerators. We will then review the current literature on the human exposure and health effects from waste incineration.

A3.1 Risk assessment

Ordinarily risk assessment for new developments is part of a HIA. There is little dispute about what this is, for example:

health impact assessment is a practical approach used to judge the potential health effects of a policy, program or project on a population, particularly on vulnerable or disadvantaged groups. Recommendations are produced for decision-makers and stakeholders, with the aim of maximising the proposals positive health effects and minimising the negative health effects (World Health Organization, 2007).

This is one of several common definitions of HIA, and while useful, does not show the extent to which the process can be contested in practice. Every part of a HIA poses substantial challenges to policy-makers, practitioners and the various stakeholders. Many of these challenges arise from genuine conflicts of interest between stakeholders, in particular, the developers, and the local community. Both sides in such conflicts may turn to “science” to bolster their respective cases.

Scientific evidence is only one part of a HIA, and often not the determining part. There are many practical challenges in doing HIA, and often deciding on the boundaries of the process in a particular case is the most difficult. A major benefit of HIA is making all participants aware of the complexity of the impacts, and of the uncertainty of the consequences of a new development. HIA, if done well, is likely to provide as many new questions as answers.

A3.1.1 Hazards associated with waste management

Hazards from waste management can be thought of as those inherent to the waste, transport-related, produced as a result of the management option, and those disseminated as a result of the management option. Most attention in work on incineration has focused on stack emissions, but these may not be the most important hazards in a particular setting.

For example, with hazardous chemical wastes, say contaminated organic solvents, there are obvious inherent risks in the material itself, wherever it is kept. Transport adds the risk of unexpected large-scale contamination, in a road or train accident say, and the emissions from the means of transport chosen. Hazards produced by incineration may be very modest, especially if the plant is well run, but must be considered. Finally stack emissions might include whatever was contaminating the solvents, as well as combustion products such as CO₂, and dioxins.

A3.1.2 Emissions to the environment from municipal waste incineration

Stack emissions are the most visually obvious emissions from waste incinerators, and the tall chimneys serve to draw the eye, and focus public concern. Emissions include water, CO₂, SO_x, NO_x, particles covering the full range of sizes – PM₁₀, PM₅, PM_{2.5} and ultrafine, organic compounds including PAH, dioxins, furans and other halo-carbon compounds, metals and others (Crowley et al., 2003).

Ash, the solid bulk residue from incineration, may pose more of a problem, and can certainly be more hazardous than the original municipal waste, unless properly managed, and safely disposed of. It is usually divided into fly ash, and bottom ash. Fly ash is collected from the exhaust gases, whereas bottom ash is more solid material left at the bottom of the grate after burning. Both types of ash need to be handled carefully, and may be categorised as hazardous waste depending on what waste materials are taken into the plant for processing.

Other emissions which need to be considered include noise, odour, wind blown waste, pest species attracted to the site, transport-related emissions, and emissions of dusts and spore from the waste material.

Other impacts from the operation of a plant might include sleep disturbance, traffic congestion, traffic hazards, light, habitat damage, stream pollution, run-off from waste storage areas, and traffic movement areas, and spills from chemical and fuel storage areas.

A3.1.3 Assessing the risk

Risk assessment poses a number of specific problems. Assessing human exposure from the operations is difficult, both because of the sheer diversity of possible emissions, but also because there are many other sources of each of these. It is seldom possible to say that a specific contaminant, either in the environment, or in people, comes from an incinerator, rather than some other source. For some exposures it is possible to use biomarker studies, to estimate personal exposure, but for many this cannot be done. In most cases it is not possible to distinguish between emissions from incinerators, and other industrial sources, although in the, relatively rare, situation when a waste incinerator is built in an otherwise rural environment, this may be possible.

The model of exposure used in many early studies – relating exposure directly to distance from the chimney, is inadequate (Lloyd, 2004). For example Fierens et al. (2007) found the main route of dioxin exposure in a small Belgian study was by eating meat raised on farms near an incinerator chimney, and Huang et al. (2007) found that environmental dioxin levels were related to predictions from atmospheric dispersion models around 19 municipal waste incinerators, but that serum dioxin levels in the population were not. Gonzalez et al. (2000) did a study before and two years after the start of a new municipal incinerator in Spain, and found that dioxin levels rose equally in people living close to and far away from it.

Assessing the possible consequences of exposure is far harder, for at least two reasons, confounding and the low absolute risk expected. By confounding we simply mean that for any possible outcome from exposure to waste incinerator emissions, there are other causes, and some of these causes will be, or will appear to be, associated with exposure to the emissions. Given the relatively crude exposure assessments common in these studies, a major source of confounding is the pattern of human settlement, and its development over time. Many waste facilities are built in industrialized areas, and the adjoining housing tends to be used by poorer people, at least some of whom work in the local industries. All of these factors, exposure to other sites, social deprivation, and occupational exposure, are potent confounders for the exposures considered here.

The likely effect of exposure to waste incineration is modest. For example, it is very unlikely that the risk of any adverse event would be as much as doubled by proximity to even the most ineptly managed waste plant. More realistic estimates would be relative risks of the order of 1.1 to 1.3. These effects are hard to identify in most studies. Studies of single-sites face a particular challenge, as the number of people exposed is likely to be small. The power of any epidemiological study to detect these types of risk is very limited (Tango, 2002, for example). An implication of this is that the failure to demonstrate a hazard, does not suffice to show that an exposure is safe. In particular policy-makers should put more weight on toxicological studies, and environmental dispersion studies, than on negative epidemiological studies (Crowley et al., 2003).

A further practical issue, but one of great policy importance, is that the majority of published studies relate to older plants. With the more recent European Union regulations, (European Commission, 2000) many older plants have closed, or been fitted with much more stringent emission controls. While this is obviously desirable from a public health perspective, it does pose the difficult question, of the relevance of studies around older plants, to populations affected by more modern facilities.

A3.2 Health effects

Existing studies of the health effects of emissions from municipal waste incinerators have concentrated on local health effects in three areas:

1. respiratory symptoms and illness
2. reproductive effects, especially congenital anomalies
3. cancer.

The long-term health impact of the CO₂ emissions have not been considered in detail.

A3.2.1 Respiratory signs and symptoms

A surprisingly small number of studies have addressed this question so far. Waste incineration is known to produce significant quantities of small airborne particles, indeed these particles make up the “fly ash” collected from such facilities, and on modern plants there are filtration systems to remove as much as possible of these from the exhaust gases. The health effects of fine airborne particles are now known to be of the highest public health importance. It is less clear whether living near a waste incinerator leads to any specific additional risks over and above those from exposure to other sources of particulate air pollution.

Shy et al. (1995) studied three separate American populations (in North Carolina) living near a biomedical incinerator, a municipal incinerator and a hazardous waste incinerator. The investigators measured air quality, respiratory symptoms and respiratory function in these populations. Results were compared with three matched-comparison communities. No differences in concentrations of particulates were detected among the three pairs of communities. For the municipal incinerator, it was reported that emissions accounted for 2% of the fine particulate mass detected at the monitoring station. Symptoms of respiratory illness, such as chronic cough, wheeze and sinus trouble, were significantly greater in those living near the hazardous waste incinerator than in their control community. However, this difference did not remain when all three incinerators were combined and compared with their comparison populations. Further analyses of the same study (Hazucha et al., 2002, Lee & Shy, 1999) showed no difference in respiratory symptoms or pulmonary function between study participants from communities with an incinerator and those without.

In another cross-sectional study, the frequency of respiratory symptoms was examined in children living near two sewage-treatment facilities with high-temperature sludge-burning incinerators in Sydney (Gray et al., 1994). The results of lung function tests and the prevalence of asthma, symptom frequency and atopy were not significantly different between the study and control populations. Socioeconomic status, however, was not taken into account when comparing the two regions, nor was parental cigarette smoking, nor indoor air quality.

A3.2.2 Reproductive effects

The main reproductive effect studied is twinning. While the available studies are not consistent, nor conclusive, there is at least an indication of an excess of twin births amongst humans in areas which are industrialized and areas which may be exposed to emissions from waste incinerators. For example, Obi-Osius et al. (2004) showed an excess of twin births in heavily industrialized areas of Hesse, and in women living near a toxic waste incinerator. Studies in central Scotland (Lloyd et al., 1988, Williams, F. L. R., Lawson & Lloyd, 1992) showed a similar excess of twin births and an excess of female births in women living near waste incineration facilities. On the other hand Rydhstroem (1998) in a Swedish study, using routinely collected data, found no evidence for an excess of twinning in parishes containing waste incinerators. This is an area which needs further systematic study.

A3.2.3 Congenital anomalies

There are several recent studies of this question. Cresswell et al. (2003) found possible evidence for an increased risk of all congenital anomalies close to a large plant in the north of England. Tango et al. (2004) found possible evidence for an increased risk of a range of adverse pregnancy outcomes, especially death due to congenital anomalies, between 1 km and 2 km, from 63 high dioxin emitting waste municipal incinerators in Japan. Cordier et al. (2004) showed no overall increase for congenital anomalies around municipal waste incinerators in France, but did find an excess of oro-facial clefts and renal dysplasia. They also identified a link between congenital anomalies and road traffic density. Dummer et al. (2003) identified an excess risk of lethal congenital anomalies especially spina bifida and cardiac defects around incinerators, and anencephaly and still births around crematoria. Nouwen et al. (2001) studied a small population in one area of Flanders, close to an incinerator, and found a suggestion of an increased risk of congenital anomalies in this group. Ten Tusscher et al. (2000) studied the consequences of exposure to dioxins from a Dutch incinerator in the 1960s and found an increased risk of oro-facial clefts in the exposed area.

Overall these studies raise more questions than they provide answers. There is certainly some evidence for adverse health effects, and this is most clear in the studies around the oldest facilities. This is an area that requires further studies, but it would be rash to acquit incinerators on the basis of this evidence.

A3.2.4 Cancer

A series of studies, including both case-control studies and ecological studies, by Viel and his colleagues of the community around a waste incinerator at Besançon (Floret et al., 2007, Floret et al., 2004, Viel et al., 2000) have shown that there is an increased risk of STS in the area, that this risk is highest in the zone where exposure to the incinerator is estimated to be highest, that living in this zone is an important risk factor for sarcoma, and the incinerator is the principal source of exposure to dioxins and furans in the area. Viel and his colleagues have also

shown suggestive evidence of a risk of NHL (but not Hodgkin's disease) in the same area (Floret et al., 2003, Viel et al., 2000).

These results for sarcoma are confirmed by Zambon et al. (2007), in a case-control study which shows a significantly increased risk, which increased with both duration of exposure, and modelled intensity of exposure to incinerator emissions, amongst people living in Venice and Mestre. This is, we think, the most convincing demonstration of an increased risk of cancer associated with residence near waste incinerators in the literature, but note that Tessari et al. (2006) in an ecological study of much of the same area as Zambon et al. (2007) found little evidence of increased risk. This may reflect the limited power of spatial studies to detect significant real risks.

Elliott et al. (2000, 1996) studied cancer incidence in the United Kingdom population living near municipal waste incinerators. Their first study showed a general increase in cancer risk, most notably for liver cancer. They felt that the generalized risk probably related to uncontrolled confounding, but that the liver cancer risk needed further study. Their second study (Elliott et al., 2000) followed a detailed evaluation of all the cases identified by the first, and confirmed an excess of liver cancer in the study areas, although this may still be, at least partly, related to confounding by deprivation.

In an analysis of childhood cancers, Knox (2000) examined migration patterns around 70 municipal waste incinerators, 460 landfill sites and 307 hospital incinerators. Although there were no significant association related to landfill sites, there was a highly significant excess of migration away from birthplaces close to incinerator sites. The author comments that these findings may be the result of age-related circulation around available housing stock, with, for example, young mothers living with their inner-city parents and moving out to less industrial areas over time.

Elliott et al. (1992) analyzed the incidence of cancers of the larynx and lung near an incinerator of waste solvents and oils at Charnock Richard in Lancashire (which operated between 1972 and 1980) and nine other similar incinerators in Great Britain, after reports of a cluster of cases of cancer of the larynx near Charnock Richard. No increase in cases of cancer of the larynx was detected near these sites.

Michelozzi et al. (1998) analyzed deaths from cancer in a densely populated suburb of Rome with multiple sources of pollution, including a large waste disposal site, an oil refinery plant, and a waste incinerator. Cancer risks were estimated for increasing distances from the plants. No excesses of cancer were found in the study area. However, a significant finding was that the risk for cancer of the larynx in men decreased with increasing distance from the plant. Kidney cancer in women living between three kilometres and eight k of the plants was also found to be increased. This, however, was not influenced by increasing distance from the site. Factors such as deprivation were taken into account in this analysis, as zones near industrial areas are generally socioeconomically disadvantaged.

Biggeri et al. (1996) did a case-control study of lung cancer deaths around four sources of environmental pollution (shipyard, iron foundry, incinerator, and city centre) in Trieste, Italy. The risk of lung cancer was strongly associated with residence near the city centre and near the incinerator. In each of these two locations, as distance increased from the source, risk was reduced. The observed effects in relation to the city centre may have been influenced by the close proximity of two of the other sites, namely the shipyard and, to a lesser degree, the iron foundry. Biggeri and Catelan (2005) showed increased death rates from NHL in males, but not females, living in a municipality (Campi Bisenzio in Tuscany) with a large waste incinerator which was closed in 1986. There were too few cases of sarcoma in this study for analysis.

Parodi et al. (2004, 2005) showed an increased risk of lung cancer death in females living near industrial sites – a coke oven (Cornigliano, Genoa), and an industrial complex which included a waste incinerator in northern Italy (La Spezia). There was no identified excess in males, probably because of severe confounding by smoking status and occupational exposure.

Comba et al. (2003) did a case-control study of STS in Mantua. This is a middle-sized city in Italy, from where a cluster of cases of STS had previously been reported, and where a large hazardous waste incinerator had operated until 1991. They found a significant association between residence close to the site, and risk of sarcoma.

Fukuda et al. (2003) studied municipality level mortality data from Japan. They compared municipalities with an incinerator to those without, and found no statistically significant difference in mortality from any major cause, once an adjustment for socioeconomic deprivation was made.

An overall interpretation of this body of literature is not simple. Each of these studies has limitations, and in each case, except the Besançon studies, alternative explanations for the observed excess risk can be found, either other sources of environmental contamination, or residual confounding by social deprivation, or both. Taken all in all it would be prudent to acknowledge the very strong likelihood that residence in contaminated industrial areas increases individual risk of cancer and that it is certainly possible, and in fact probable, that waste incinerators can contribute to this increased risk. It is not so clear what the implications of this are for modern plants.

A3.3 Health effects of incineration

For the reasons discussed above, it is likely to be very hard to conduct epidemiological studies which will clearly identify health risks associated with the incineration of waste, especially around plants which comply with the EU regulations. A point which should be made is this – if waste incinerators produce emissions known to be harmful, there is no reason to suppose that these will be any less (or more) harmful than the same emissions from other sources. Although, for example, waste plants produce only a small fraction of the total particulate air pollution in most places, that small fraction is definitely harmful.

In the light of the evidence briefly reviewed above, it is reasonable to believe that waste incinerators have contributed at some level to the total burden of disease from environmental pollution in Europe and the United States. Waste incinerators of the designs studied are now banned in wealthy countries, but are still being constructed in poorer countries, so this data is of more than historical interest. It is not likely that these incinerators have played a predominant role in many places, but they have, almost surely, caused harm to surrounding communities.

Two questions arise.

First, are the health effects of older incinerators well characterized? The answer to this is definitely no – several health effects have hardly been studied at all, and there are many unanswered questions in the existing literature. Some of these are briefly discussed above.

The second is – what are the implications of these data for newly built plants in wealthy countries? It is argued that as these facilities produce far lower levels of pollution than older plants, then health concerns are irrelevant when making policy on their construction and operation. This is true, to a point, but only to a point. Like other activities waste incineration can be made to seem safer by dispersing the consequences. Every phase of the construction, operation, and final disposal of modern waste facilities needs to be subjected to thorough analysis, to identify possible hazards, and either remove them, minimize them, or decide explicitly, that they are to be accepted.

A3.4 Making policy

This is not easy. Modern societies need effective waste management strategies. It is important to note that waste disposal can be very profitable, and to ensure that investment in costly landfill sites and modern incinerators does not displace investment in more fundamental measures, the “reduce, re-use, recycle” strategies of sustainable development.

The known risks of many common activities are higher than any likely effects of waste management. Passive smoking and air pollution from traffic are two important examples. However most developed countries are taking forceful action to reduce both of these. Major uncertainties will persist for at least the next decade, about the health effects of more modern methods of incineration.

In this situation the precautionary principle might be invoked: “Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation” (United Nations Conference on Environment and Development, 1992).

As discussed in a recent editorial by Martuzzi (2007) and two accompanying pieces by Kriebel (2007, positive) and Goldstein (2007, critical), this principle does not resolve the scientific uncertainty, but it might form a valuable basis for policy-makers on which to make decisions. Goldstein addresses some of the downsides of this position. Scientists can decline to make decisions pending the availability of new evidence, but legislative and administrative decisions are often made to fixed timetables.

A3.5 Policy implications

In short, the current state of knowledge is inconclusive. There is reasonable evidence of some adverse health effects from older plants. There is little or no relevant data from more modern plants. Future studies will be harder to do, and even harder to interpret, than older studies, as the levels of emissions are likely to be lower. Epidemiology may not be the most suitable tool, as it is unlikely to be possible to reduce the uncertainties fast enough for current policy needs.

A reasonable basis for future action might be this. Decisions to proceed with the construction of new plants should observe the precautionary principle, and each plant and each site should be rigorously evaluated for potential harm. There is no present evidence of harm from modern facilities, but it will be necessary to do some systematic large-scale studies, focusing on the health effects identified so far, to monitor the impact of present and future industrial activities across Europe. Every effort should be made to support less developed countries in upgrading and/or phasing out unsatisfactory facilities.

Acknowledgements

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A4. Technology of waste management and exposure assessment: landfills in Italy

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A4.1 Introduction

In 2005, the overall MSW production in Italy was approximately 31.7 Mtons (metric), with a 5.5% increase relative to 2003 and an average annual production per person of 0.53 tons. Of the total amount, 24.3% was recovered by differentiated collection and recycled, 12.1% was processed in MSWI plants with energy recovery, and some 50% was disposed of in landfills. The residual to 100% is: composted, temporary stocked (Campania region), processed in other plants for energy recovery (Italian Agency for the Protection of the Environment and for technical Services (APAT) & Waste National Observatory, 2007).

The data on the municipal waste management in 2005 confirm a reduction of landfill disposal compared with the years before (-2.7% compared to 2004) (Italian Agency for the Protection of the Environment and for technical Services (APAT) & Waste National Observatory, 2007).

In 2005, landfill plants for municipal waste in Italy were 340, whereas in 2002 were 552: so in the last three years in Italy more than 200 landfill plants were closed, because other disposal techniques (incineration, composting) were more used and, at the same time, the differentiated collection and recovery, in the last years, is more encouraged (Italian Agency for the Protection of the Environment and for technical Services (APAT) & Waste National Observatory, 2007).

In 2004, the special/industrial waste production in Italy was 108.4 Mtons. Of the total amount, 56.5 Mtons were special non-hazardous waste, 5.3 Mtons were special hazardous waste, 46.5 Mtons were construction/demolition waste, with a 17.7% increase for the special/industrial waste and a 14.3% increase for the hazardous waste, relative to 2002–2004 (Italian Agency for the Protection of the Environment and for technical Services (APAT) & Waste National Observatory, 2007).

The special/industrial waste management shows what follows:

- 54.1% is processed for matter recovery
- 23.9% is disposed of in landfills
- 17% is processed in chemical–physical or biological treatment plants
- 3.6% is used as energy source
- 1.4% is incinerated.

Therefore, approximately 1 Mton of special hazardous waste was disposed of in landfill plants. Precisely, the landfills for special/industrial waste registered in 2004 are so divided: 497 landfills for construction/demolition waste, 127 landfills for special non-hazardous waste, 6 landfills for hazardous waste (Italian Agency for the Protection of the Environment and for technical Services (APAT) & Waste National Observatory, 2007).

A4.2 Italian regulation about landfills

The Legislative Decree n. 36/2003, acknowledging the Directive 1999/31/EC, is the present Italian regulations about waste landfills (Decreto Legislativo 13 gennaio 2003, n. 36 “Attuazione della direttiva 1999/31/CE relativa alle discariche di rifiuti” [Legislative Decree 13 January 2003, n. 36 "Implementation of 1999/31/CE Directive relating to waste landfills", Supplemento Gazzetta Ufficiale Italiana n. 59 12/03/2003 (in Italian)). This Decree, as the EU Directive, is based on the following criteria:

- to reduce the quantity of biodegradable waste to landfill assigned
- to reduce the quantity and the toxicity of waste to landfill assigned
- to define the planning and working rules for the new and the operative landfills
- to encourage the previous treatment of waste before disposing of in landfill

- to avoid the mixture of potentially hazardous waste, by considering that some waste typologies can be disposed of only in particular sites.

The planning and working criteria provided in the Legislative Decree 36/2003 for the new and the operative landfills are very strict and provide a soil and water protection and a post-closing management for 30 years at least. Unfortunately, in Italy, many landfill sites were set up before the Decree 36/2003, so, they often have not the suitable environmental protection systems; besides, especially in some areas of southern Italy, municipal, special and hazardous waste are disposed of in abusive landfills. This practice is dangerous for the human and environmental health.

A4.3 Emission of hazardous and non-hazardous substances from landfill sites

In general, a landfill site can have an impact on all environmental compartments. In particular, a landfill can have the following effects on (European Commission, 1999):

- air: emissions of methane (CH₄), CO₂; smell; volatile organic substances;
- water: leaching of salts, heavy metals, biodegradable organic and in groundwater persistent substances;
- soil: accumulation of potentially hazardous substances;
- landscape: soil occupation, restriction of other uses;
- ecosystems: contamination and accumulation of toxic substances in the food chain; and
- municipal areas: potential exposure of population to hazardous substances.

Some national (carried out also by the National Institute of Health) and international studies found in groundwaters near landfills the substances reported in Table A1 (Bellino et al., 1999).

Concentration range of substances found in the well down to hazardous landfills depends on the type of dumped waste.

A4.4 Exposure assessment

The population is more and more worried about a possible health risk related to waste disposal plants. Therefore, it is necessary to carry out studies aimed at identifying a possible cause–effect relationship between waste disposal systems and state of health of resident populations near disposal sites. To this end it is necessary:

- an environmental characterization to identify possible risk factors (landfills, industries, agricultural activities, motorvehicle traffic, etc.) and a possible “impact area”;
- an exposure assessment; and
- epidemiological studies.

In many epidemiological studies on the health effects from waste, the exposure assessment in resident populations is based on the distance between the waste disposal/treatment sites and possible exposure areas for population groups. In the last years, in some studies, the authors considered a distance of 2–4 km between the exposure areas and landfill site. Jarup et al. (2002) used buffers of 2 km around the landfills (this distance is considered the dispersion limit of the emissions in air and water) while Dolk et al. (1998) and Fielder (Fielder et al., 2000) used buffers respectively of 3 km and 4 km. Jarup et al. (2002) do not think that a finer resolution (for example 1 km) can be more informative, as discussed also by Elliot et al. (2001a). Similar choices were made by Morris et al. (2003) in Scotland and by Boyle et al. (2004) in the Republic of Ireland who adopted 2–3 km circles and did not use 1 km circles, because of large data fluctuations. In 2004, a study, coordinated by the Istituto Superiore di Sanità (ISS) (Bellino et al., 2004), on the assessment of the exposure and the health impact for residents near landfills was issued. In this paper, a distance of 5 km between the exposure areas and landfill site was considered, in order to get a more “powerful” study (a larger numerosness of the potentially exposed population).

A4.5 Conclusions

Studies on the risk related to landfills are necessary to develop and implement, at the national level, a specific “protocol”, in order to assess the health risk for residents near impact areas of landfills emissions. This protocol should provide also an integration of the environmental and health studies.

TableA1. Concentration range of substances found in the well down to the urban landfills

PARAMETER	CONCENTRATION RANGE	PARAMETER	CONCENTRATION RANGE
Chloride	50–250 mg/l	Selenium	1–10 µg/l
Nitrite	20–130 mg/l	Zinc	800–3000 µg/l
Nitrate	5–10 mg/l	Benzo (A) Anthracene	0.008–0.1 µg/l
Sulphates	80–250 mg/l	Benzo (A) Pyrene	0.006–0.01 µg/l
Iron	0.5–10000 mg/l	Benzo (A) Fluoranthene	<0.005–0.1 µg/l
Potassium	0.1–1 mg/l	Benzo (K) Fluoranthene	<0.005–0.05 µg/l
Magnesium	2–45 mg/l	Benzo (G, H, I) Perylene	0.008–0.1 µg/l
Manganese	0.1–2000 mg/l	Chrysene	0.02–5 µg/l
Arsenic	1.5–10 µg/l	PAHs	0.01–0.1 µg/l
Barium	10–2000 µg/l	Benzene	0.03–1 µg/l
Beryllium	3–4 µg/l	Toulene	<0.01–15 µg/l
Boron	80–1000 µg/l	Ethylbenzene	<0.01–50 µg/l
Cadmium	<0.1–0.5 µg/l	P-Xilene	<0.01–10 µg/l
Cyanide	<20–50 µg/l	Styrene	<0.01–25 µg/l
Vinyl Chlorade	<0.01–0.5 µg/l	Aliphatic halogenated	
Chromium tot	1–50 µg/l	Hydrocarbons (tot)	0.08–10 µg/l
Chromium VI	<2–5 µg/l	Monochloro Benzene	<0.005–40 µg/l
Mercury	<0.1–1 µg/l	1, 2 Dichlorobenzene	<0.005–270 µg/l
Nickel	3–20 µg/l	1, 4 Dichlorobenzene	<0.005–0.5 µg/l
Lead	0.3–10 µg/l	1, 2, 4 Trichlorobenzene	<0.005–190 µg/l
Copper	35–1000 µg/l		

A5. Technology of waste management and exposure assessment: incinerators

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A5.1 Introduction

In Italy the annual amount (2005) of municipal solid waste processed in incineration plants is approximately 12% of total production. Waste incineration techniques, with no recycling or forms of energy recovery, were originally used for reasons of hygiene and volume and weight reduction. Nowadays, incineration is possible only if associated with energy recovery; in general, in addition to crude MSW, some treated MSW comes as a residual from differentiated waste collection and/or as a refuse-derived fuel (RDF).

According to recent surveys (Italian Agency for the Protection of the Environment and for technical Services (APAT) & Waste National Observatory, 2007, Italian National Agency for New Technologies Energy and the Environment (ENEA) & Federambiente, 2006), 51 MSWIs are presently active in Italy and treat approximately 4.4 Mtons/year of MSW together with clinical waste and RDF. Geographically, they are primarily located in the northern regions (30 plants, 60%), providing the major amount of waste incineration (3.4 Mtons, 77%) and energy recovery (2.1 GWh electrical, 80%; 0.71 GWh thermal, 100%). Per cent estimates are relative to the corresponding total nationwide figures.

Several other surveys show the evolution of waste incineration in the last twenty years (Caggiano, Cipriano & Viselli, 2003, Federambiente, 2001, Viviano, 1988, Viviano & Ziemacki, 1987): dismissal of old and obsolete plants and an increasing percentage of incinerated waste by new plants with a higher potentiality, according to the BAT and with energy recovery.

A5.2 Italian regulations and guidelines

Already with national Legislative Decree 05.02.97 No. 22 “Implementation of Directives 91/156/CEE on waste, 91/689/CEE on hazardous waste, and 94/62/CE on packaging and packaging waste”, the Italian legislation pointed at combustion of waste and RDF, with energy recovery, as one of the “recovery operations” (De Stefanis, 1998).

- Directive IPPC 96/61/CE, fully absorbed by national Legislative Decree 18.02.05 No. 59, defined the BAT concept, while the recent Integrated Pollution Prevention and Control (IPPC) BAT Reference (BRef) Bureau works (Sevilla) set out the BRef also for incineration plants (European Commission, 2006). The EU BRef has been taken up by the national BRef “guidelines to identify and employ the BAT for incineration plants”.
- Lastly, national Legislative Decree 11.05.05 No. 133 “Implementation of Directive 2000/76/CE on the matter of waste incineration” enacted the last dispositions concerning incineration plants (Commission ex Art. 3 Comma 2 DLivo 372/99, 2003).

A5.3 Incineration technology and emissions

Emissions of waste (MSW or RDF) combustion are similar to those of other solid fuels and mainly consist of:

- gases such as carbon oxides (CO and CO₂) and their acid products, NO_x, SO_x, HCl, water vapour;
- non-combustible matter and products of incomplete combustion such as particulate matter and/or vapours, including silicates, inorganic ash, soot, metal elements and their oxides and salts (for example, mercury and other metals with high vapour pressure); and
- organic emissions comprise priority pollutants, namely PCDDs and PCDFs and PAHs (National Research Council, 2000).

The different types of thermal treatment techniques and flue-gas cleaning systems applied in incineration plants in Italy are as follows:

- Thermal treatment techniques
- grate incinerators (83% of total MSW incinerated)
- fluidized bed incinerators (15% of total MSW incinerated)

- rotary kiln incinerators (2% of total MSW incinerated).
- Flue-gas multistage cleaning systems
- dust removal: electrostatic precipitator, bag filters;
- NO_x reduction (deNO_x): selective non-catalytic and catalytic reduction processes;
- acid gas removal: dry, wet and semi-wet scrubbing systems;
- PCDD and PCDF reduction: primary measures (that is, combustion), secondary measures (that is, abatement by catalysis, adsorption); and
- mercury reduction: wet scrubbing, adsorption, condensation.

Indicative values of PCDD+PCDF releases to air from some old MSW plants in the 1980s and over the 1990–1991 period appear to be in the ranges of 2–60 ngI-TEQ³/Nm³ (highest emission: 2000 ngI-TEQ/Nm³) (World Health Organization, 1987) and 0.04–80 ng I-TEQ/Nm³ (Hutzinger & Fiedler, 1996), respectively. For selected pollutants, the emission ranges (annual averages) for some modern European MSW plant are summarized in Table A2, with best emission performances being related to BAT application (European Commission, 2006).

Table A2. Annual average emission ranges of modern European MSW plants

DUST mg/m ³	HCl mg/m ³	NO _x mg/m ³	Hg mg/m ³	Cd+Tl mg/m ³	OTHER METALS ¹ mg/m ³	Σ(PCBs) mg/m ³	PAHs mg/m ³	PCDDs+ PCDFs ² ng/m ³
0.1–4	0.1–6	20–180	0.0002–0.05	0.0002–0.03	0.0002–0.05	<0.005	<0.01	0.0002– 0.08

Note: ¹Σ(Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V); ²Calculated as I-TEQs of 2,3,7,8-T₄CDD.

A5.4 Case study: the Melfi incinerator

In Italy, all incineration plants are equipped with a sequence of three to five cleaning systems to minimize pollutant emission; 22% of the plants have also a double-stage dust removal and 86% a deNO_x stage. Normally, incinerators comply with emission limits and the emission factors (pollutant per ton of waste incinerated) are related to the air pollution control system applied.

In general, the stack is at least 70 metres high; in large modern plants, stacks are higher (for example, Brescia and Acerra incinerators have stacks respectively as high as 120 metres and 110 metres). Effective stack height (geometric plus enthalpic thrust), local atmospheric conditions and topography situation are decisive for dilution of emissions (in general, greater than 10⁴–10⁶ at the point of maximum fall-out 1–10 km downstream). The contribution of microcontaminants to their concentrations in ambient air is consequently in the order of ng/m³ for the particulate matter, less than pg/m³ for heavy metals, and well below fg/m³ for PCDDs+PCDFs (as I-TEQs).

The ISS and the Regional Agency for Environmental Protection of Basilicata Region (ARPAB) are carrying out a study to investigate the local environmental impact of an incineration plant (Bove et al., 2005). This plant, called “Fenice”, is equipped with two combustion lines to process MSW, chemical and hazardous waste: a grate incinerator with a rated capacity of 100 tons/day and a rotary kiln incinerator with a rated capacity of 150 tons/day. The plant is located within a small industrial area, near the city of San Nicola di Melfi (Potenza) in southern Italy. For the study, many samples of plant stake emissions, ambient air, topsoil and local farm products are being collected and analyzed. Below, some emission (Table A3) and environmental contamination data obtained during 2003–2004 have been summarized.

- PCDD and PCDF deposition rates: throughout the area explored, daily deposition rates of PCDDs+PCDFs cover the rather narrow range of 35–55 pg/m² or 1.5–2.3 pgWHO-TE/m² (cumulative values are expressed in analytical units and World Health Organization Toxic Equivalent (TEQs, in this case indistinguishable from the parallel I-TEQs). These values, detected in a winter period at six sites exposed to different emission

³ International toxic equivalent

sources and distant kilometres from one another, are among the lowest findings in Italy and comparable to deposition values found in rural areas in the EU (Carrera et al., 2002, Fiedler et al., 1999).

- PCDDs and PCDFs in PM₁₀: PCDD+PCDF concentrations in air (86–150 fg/m³ or 2.7–3.2 fgWHO-TE/m³) were obtained from PM₁₀ high-volume samplings at three sites located at approximately a 1-, 2- and 4-km distance around the incineration plant. These winter atmospheric concentrations are significantly lower than those measured in urban locations such as Florence and Rome and are in line with the PCDD+PCDF levels measured in air at open range sites such as the Simbruini national park near Rome (41–160 fg/m³ or 2.1–6.6 fgWHO-TE/m³) or the Castagneto hills near Florence (140–270 fg/m³ or 2.6–10 fgWHO-TE/m³), both in the Apennines mountain range (Berlincioni et al., 2002, Turrio-Baldassarri et al., 2001). Based on the amounts of PM₁₀ particulate matter collected, the PCDD+PCDF concentrations on such matrix may be estimated in the order of 200–300 pgWHO-TE/g.

Table A3. Synopsis of “Fenice” incineration plant emission data obtained during 2003–2004

STAKE EMISSIONS (11% O ₂)	ROTARY KILN		GRATE	
	AVERAGE (NO. OF SAMPLINGS)	RANGE	AVERAGE (NO. OF SAMPLINGS)	RANGE
Flue gas (m ³ /h)	71 960 (6)	67 860–80 600	66 064 (3)	59 440–76 230
Dust (mg/Nm ³)	2.8 (3)	2.0–3.6	2.4 (2)	1.7–3.1
Hg (mg/Nm ³)	0.008 (4)	0.0004–0.015	0.034 (3)	0.020–0.046
Cd (mg/Nm ³)	0.005 (5)	0.001–0.012	0.004 (3)	0.003–0.006
PCDDs+PCDFs (ng I-TEQ/Nm ³)	0.0060 (3)	0.0022–0.0110	0.0046 (2)	0.0025–0.0067

Acknowledgements

The study carried out on the Melfi incinerator has relied on the following co-workers: Bruno Bove (ARPAB); Giorgio Cattani, Maria Carmela Cusano, Silvia De Luca, Elena Dellatte, Igor Fochi, Anna Rita Fulgenzi, Nicola Iacovella, Marco Inglessis and Gaetano Settimo (ISS).

A6. Sustainable waste management in the UK: the public health role

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A6.1 Introduction

The most common definition of sustainable development is that by the Brundtland commission, that is “development that meets the needs of the present without compromising the ability of future generations to meet their own needs”. Managing waste sites in a manner that minimizes toxic impacts on the current and future generations is obviously a crucial part of this. Although the management of waste facilities is extremely complex, the IPPC regime, which requires the input of public health professionals on the regulation of such sites, means that all waste management installations should now be operating in a fashion that minimizes any immediate toxicological risks to human health.

However, the impacts upon climate change, resource use and health inequalities, as well as the effects of waste transportation, are currently not considered to be part of public health professionals’ responsibilities when dealing with these sites in the United Kingdom. Also in the United Kingdom there is currently no requirement for public health professionals to become involved in waste management planning issues and a recent study showed that public health professionals are rarely involved in waste management issues (Jordan, 2007).

A6.2 Direct toxicological impacts upon health

Defining sustainable development in terms of the quality of life of current and future generations clearly means it is important to ensure that waste management sites pose minimum risk to health. Concern about the health effects of waste installations has resulted in many epidemiological studies around these sites (Vrijheid, 2000). These have reached different conclusions, although several well publicized studies have suggested the possibility of a link between landfill and adverse health effects.

These studies are limited by a lack of accurate exposure assessment, the influence of confounding factors and reporting bias (Committee on Toxicity, 2001, Department of Health, 1998, Enviro Consulting Ltd et al., 2004). Risk assessments of these sites using exposure and toxicological data; have indicated that well-managed landfill sites are unlikely to cause adverse health effects (Enviro Consulting Ltd et al., 2004). These risk assessments, however, usually only adequately consider risks from inhaling atmospheric pollutants, as other exposure pathways such as soil, water and food are poorly understood (Royal Society, 2003, Vrijheid, 2000).

A frequent request from the local community around individual waste sites is for an epidemiological study. This is unlikely, however, to be helpful, because of problems with statistical power due to the small populations living close to waste sites (Dolk, 2002). In these particular cases it may be more helpful to undertake a health risk assessment of the installation, based on exposure and toxicology data. Another option is to carry out a HIA detailing all the potential impacts (either quantitatively or qualitatively) that the waste management site could have upon the health of the local community and then consider possible mitigation measures (Matthews, 2004, Scott-Samuel, Birley & Ardern, 2001).

In previous waste management incidents health professionals have successfully involved the local community in decision-making by creating steering groups in which community representatives are involved with the other relevant agencies in taking decisions regarding any health study (North Sheffield Primary Care Trust, 2003). This allows for the difficulties in assessing the impacts of waste management installations to be explained first-hand, and means that the local community know that their concerns are being taken seriously and that they are involved in decision-making.

A6.3 Regulation of waste management sites

Regulation of waste management sites has significantly improved in recent years. By 2006, all landfill sites and incinerators are required to have IPPC permits. IPPC is enforced by the Environment Agency in England and Wales, the Scottish Environmental Protection Agency in Scotland and the Environment and Heritage Service in Northern Ireland. These agencies grant permits based on emissions to all environmental media. The IPPC applicant needs to show that they are using the BAT, balancing the costs to the operator against the benefits to the environment (Chemical Hazards and Poisons Division - Health Protection Agency, 2004). Protecting human health is a core part of IPPC, reflected by the fact that Primary Care Trusts are now statutory consultees to the process. For the first time, health professionals have the opportunity to get involved in the regulation of waste management installations. Their role is, however, limited. There are no specific references to human health in IPPC applications, meaning that health professionals must often base their decisions on emissions data. IPPC only considers the direct effects of any

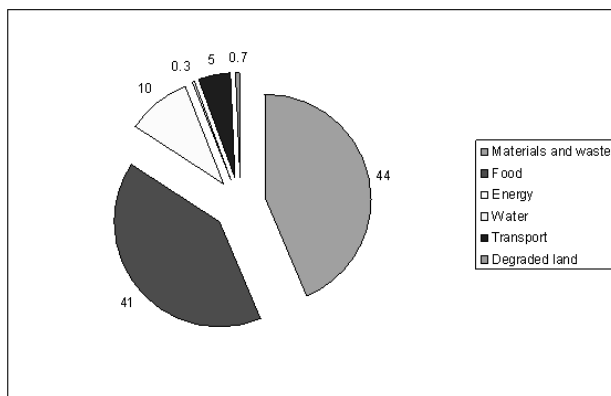
process upon health and does not allow for consideration of broader issues, for example, the effects upon traffic, resource use, climate change and health inequalities (Chemical Hazards and Poisons Division - Health Protection Agency, 2004)

Despite these limitations, IPPC is an important tool for protecting public health. Health professionals should ensure that waste management installations operate to the highest standards possible to protect public health. This could be done by discussing the IPPC application with the regulator and ensuring that the techniques used are appropriate and that there is appropriate environmental monitoring carried out around the site.

A6.4 Indirect impacts upon health: the impact of waste management on resource use and climate change and how public health professionals can help

Waste disposal is a direct result of resource use. Current populations are using up resources (including the ability of the earth's ecosystems to absorb pollution) at an unsustainable rate. The United Kingdom's current resource use is such that its ecological footprint (the area of the earth's surface needed to provide the materials and energy used without drawing on non-renewable resources) is currently about eight times the area of the country (Clift, 1998). Studies of ecological footprints of the United Kingdom population have put material use and waste management as the biggest factor influencing this calculation (see Figure A1) (Best foot forward Ltd, 2002). Understanding the exact implication of the measure of the ecological footprint is quite complex, although the predominant view is that the current pattern of material use and waste production is unsustainable.

Figure A1. Ecological footprint of Londoners by component



Source: Best foot forward Ltd, 2002

Inequitable resource use (nature's ecosystems' ability to deal with the waste that humans produce are also a resource) has serious adverse impacts upon health. According to the majority of the world's leading scientists, global climate change poses a major threat to society (King, 2004). Current patterns of resource consumption and subsequent waste disposal are exacerbating global climate change. Gaseous emissions from landfill sites and incinerators directly contribute to the problem. The use of landfill and incineration as waste management options may also indirectly encourage more resource use, as they give little incentive to either recycle or minimize waste.


Manufacturing goods from recycled materials can often require less energy (therefore, producing less greenhouse gases) than producing goods from virgin materials. Waste prevention is obviously even more effective at reducing emissions. Climate change is expected to have a major impact upon health, including increases in heat strokes, respiratory and cardiovascular problems, vector borne diseases and health problems caused by extreme weather events (Mc Michael, 2001).

Since the 1970s, a plethora of environmental legislation about waste management has been established both at EU and United Kingdom level. In response to EU legislation, the Government has produced a National Waste Strategy (Department for Environment Food & Rural Affairs, 2000). This sets out policies for waste management and reaffirms their commitment to meeting the standards set by the Landfill Directive by imposing statutory targets on local authorities forcing them to significantly increase recycling/composting rates.

Public health professionals can make a significant contribution at regional, county council and district level, where local authorities are required to produce waste strategies detailing how they will manage their waste over the next 10 years. This requirement provides health professionals with an opportunity to ensure that waste management plans consider health issues such as climate change and impacts upon resource use.

When determining waste management policies, decision-makers currently consider two principles. The first of these is the waste management hierarchy. This is a conceptual framework used by some for identifying the most (and least) sustainable ways of dealing with waste, which regards waste prevention as the most desirable option and disposal by landfill as the least attractive option (see Table A4).

Table A4. The waste management hierarchy

	Most favoured option	Prevention
		Minimization
		Re-use
		Recycling
		Energy Recovery
	Least favoured option	Disposal by landfill

The other important consideration is the proximity principle, requiring waste to be dealt with as close as possible to its point of production. This avoids transporting waste to other communities who did not produce it and reduces the impacts of waste transport, such as vehicle pollution and traffic accidents.

There is a general consensus that reduced waste production should always be the first objective of any waste management policy and that waste should be dealt with as close as possible to its source of production. It is, however, over simplistic to think that all waste management problems can be dealt with by adhering to the waste management hierarchy and the proximity principle. In the first instance there will always be some waste produced from every industrial process no matter how efficient it is; therefore, complete waste prevention is impossible (Clift, Doig & Finnveden, 2000). In addition, the high energy requirements of recycling means that on occasions it may make more sense to recover energy by incineration. The decision about the most sustainable means of waste management depends on a variety of factors, for example, the type of waste being disposed, transport distances, or the availability of raw materials. Integrated Waste Management is a system of waste management whereby rather than simply trying to move up the waste management hierarchy, waste is used in the most effective way in resource and environmental terms (Clift, Doig & Finnveden, 2000). This requires life-cycle assessment (LCA) to be carried out on waste management policies in order to determine the most sustainable means of waste management. LCA is the study of the environmental impacts of a product or service over their entire life-cycle, for example, from the extraction of raw materials, through to the consumption and final disposal of the product. When dealing with waste management strategies, LCA has the ability to consider impacts such as energy use and energy recovery. As such it offers an evidence based approach to waste management planning and is crucial if waste is to be managed in the most sustainable way (Clift, Doig & Finnveden, 2000).

Where possible, public health professionals should ensure that LCAs are carried out on waste management strategies so that the most sustainable option is chosen. Although LCA is a crucial tool for developing waste management strategies, it cannot be used to assess local site specific issues, for example, it can not determine if placing an incinerator in an area might exacerbate a local air quality problem, without additional detailed assessment of the impacts of transport and disposal of waste. As such LCA can only help identify the most efficient waste management strategies, but will not be able to answer questions about the health effects of an individual installation. Public health professionals need to ensure that where appropriate risk assessments or HIA approaches, are also carried out on waste management installations or strategies. A HIA of a waste management strategies is not a simple task, however, there are some excellent general guidelines available for conducting HIAs (Matthews, 2004). A good HIA of a waste management strategy should highlight all of the impacts upon health (both positive and negative) and suggest possible ways to improve the strategy (Matthews, 2004). The new SEA directive will also offer health professionals the opportunity to comment on this issue (Williams, C. & Fisher, 2007).

A6.5 Conclusions

Waste management is clearly an issue in which health professionals have a role and can make a positive difference. Public health professionals should be aware of the fact that unless their voice is heard in the sustainability debate in which waste management is an important issue, it may remain focussed on technical and economic issues.

Disclaimer

The authors acknowledge that the views expressed in this paper are their own and not necessarily those of their employing organizations.

A7. Public views on sources of knowledge for decisions about waste management

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We summarize the report of a large qualitative study carried out as part of a program of work on options for waste management in the Republic of Ireland. The purpose of this was to document and analyze the knowledge and perceptions of those promoting waste management systems, developing waste management policy, and members of the business community and general public affected by their operations. Our work was funded by the Irish Health Research Board.

A7.1 Methods

As part of a larger study, we carried out a large qualitative study in the Republic of Ireland. Fieldwork was completed in 2002. We held four focus groups with the general public, two in areas with operating waste plants, two in areas where such plants were being planned. We carried out 17 face-to-face interviews with people from 14 organisations relevant for waste management, waste production, or environmental policy. We did 12 telephone interviews with Environmental Health Officers, who have the primary role in the day-to-day enforcement of waste legislation in Ireland. We also sought submissions from the general public and received six.

A7.2 Waste policy

All informants were agreed that waste management in Ireland is currently facing a crisis. The precise nature of this crisis varied depending on the perceptions of the informant. There was virtual unanimity that landfilling of waste no longer offered a medium- to long-term solution; but there was disagreement as to the acceptability of incineration as a replacement means of waste management. The waste industry felt left out of policy formation.

“Currently the infrastructure in Ireland is not sufficient to deal with the quantities or the type of waste that’s generated in the business community in Ireland.”

“We came to the conclusion pretty rapidly that the current structures that are in place won’t deliver the infrastructure that’s necessary, we’ve got a lot of plans but we have no delivery.”

Source: Crowley et al., 2003

Members of the public participating in our studies favoured greater use of recycling and the introduction of measures to reduce the amounts of waste generated, but it was the “professional view” that such measures would only have a marginal impact in the medium term, requiring maintenance of substantial waste disposal capacity for the time being. “Professionals” tended to favour incineration as the option for this, and saw the major challenge to be increasing its acceptability to the general public.

At the root of this divergence lies a significant difference of opinion (that is, perception) in relation to the environmental and health hazards of the various options for waste management, the capability of existing structures and institutions to “police” compliance with the regulation of waste disposal and the likelihood of achieving significant change in public attitudes towards waste generation and waste disposal.

A7.3 Health and environmental impacts

It was difficult to draw out any distinction between “health” and “environmental” impacts in the responses of participants. In the case of landfilling, venting and potential leakage of gases, pests and water contamination were identified as health hazards. In the case of incineration, emissions of dioxins and disposal of waste ash were similarly mentioned.

In general, informants showed little detailed knowledge of epidemiological relationships. Specific health impacts were seldom listed. Informants frequently commented critically on the absence of local studies. There was a tendency from the representatives of the waste management industry to equate this absence of local studies with an absence of impact.

The perception of the general public was that incineration was “unpalatable”. In contrast, informants from the industrial and commercial sector tended to demonstrate a strong belief in the current state of incineration technology as a safeguard against health impact. This view was generally dependent on a rider concerning the quality of management; this view was even more strongly held about landfill. Service providers believed that the poor public perception of landfill and the consequent suspicion of incineration had their origins in the previous poor management of waste disposal sites.

Representatives of the service providers and of industry were generally optimistic that greater compliance with regulation could be achieved in the future. While it was frequently noted that the Republic of Ireland exhibited a “non-compliant” culture, it was felt that, in relation to waste management, this had to change because of pressure from the European Union.

There was little satisfaction with existing agencies and structures. There was a general ambiguity as to whether the primary responsibility for ensuring compliance should rest with central government or locally.

Local authorities were felt to be “compromised” or to behave erratically, because of the ambiguities arising from their responsibilities for waste management and public representation. Regional plans were perceived as duplicating, or being inconsistent with, local plans.

Health boards were seen as having a potentially greater role to play, with reference being made to the new Health Strategy, and its call for the wider use of health impact assessment and for health proofing of the plans of other sectors, and to the National Environmental Health Action Plan. Some ambiguity was also perceived in the role of health boards, given their parallel responsibility as managers of the large quantities of hazardous hospital waste.

A7.4 Information issues

From their respective positions and perspectives, informants agreed that the key to the resolution of existing disagreements on the future of waste management lay in the production of trustworthy and trusted information. Not unnaturally, the various parties were generally convinced that this would persuade other parties over to their own particular view. Apart from reservations about some press reports, most informants were confident that the information actually presented to the public was credible. For example, service providers were often complimentary of bodies like Greenpeace as a source of reliable information. Likewise, a number of informants rated the Internet as a valuable information source, yet professionals would argue that there is generally no “quality control” to assure that information available on the Internet is accurate and unbiased.

“There are genuine concerns (among the community) about the health implications of waste disposal due to a lack of information.”

“Waste management has now turned into a trust issue. They (the local community) don't trust anymore.”

Source: Crowley et al., 2003

The diffusion of better information was never perceived as being sufficient in itself to resolve local fears concerning the location of waste disposal facilities. It was felt that it would assist in giving the general public more ownership of the debate and in facilitating their participation.

A7.5 Challenges

A range of challenges to implementing an effective waste policy were identified, especially by the Environmental Health Officers. They found that there was a scarcity of suitable sites, and in every case known to them, there was well-organised opposition by local pressure and lobby groups. Communities and businesses were reluctant to engage

in waste minimisation/recycling other than on a small scale. On the other hand, there was a historical experience, by local communities, of poor waste management practice.

“Heretofore anything and everything went into landfill.”

“The Nimby syndrome is out there.”

“The general public have had negative experiences in the past.”

“The only opinion out there is that they don't want dumps near them.”

Source: Crowley et al., 2003

A7.6 Conclusions

Waste policy is recognized as an important problem by all members of the community. There is substantial agreement between the public and other actors, about the major issues, and in particular about the sources of information for policy-making. Trust remains a dominant problem, gaining and keeping the trust of the community will be crucial.

Acknowledgements

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A8. The INTARESE Project: health impact assessment of waste management

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A8.1 Introduction

The EU funded Integrated Assessment of Health Risks of Environmental Stressors in Europe (INTARESE) project aims to develop an assessment of the health impacts of several environmental exposures within an integrated conceptual framework. Waste is an area of high concern in Europe for several reasons.

- Overall waste volumes are growing in Europe and MSW generation has been contributing significantly to this growth.
- The number of different options available for the treatment and management of waste including prevention, minimization, recycling, energy recovery and disposal. 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.
- The various methods of waste management release a number of substances, most in small quantities and at extremely low levels. 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 by the various management technologies, 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 in populations exposed to emissions derived from waste management technologies.
- One important issue in waste management 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 (for example, Campania in southern Italy) and the environmental and health consequences have not been quantified.
- There are several stakeholders involved in waste, including industry, central/regional governments, city councils, nongovernmental organizations (NGOs), service users, private companies, citizens, scientists, and media. It should be recognized that there are several conflicting interests among the various stakeholders, for example 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 choices of waste management a very controversial area.

The overall aim of the INTARESE policy study on waste is to assess potential exposures and health effects arising from municipal solid wastes throughout their life-cycle, from generation to disposal or treatment. We will do the assessment at the country level: Italy, United Kingdom and Slovakia. It is expected that the methodology will be used to evaluate policy scenarios at a wider EU scale.

A8.2 Key elements/relationships for waste assessment

The study is evaluating the health impact of different management policies for MSW considering a baseline scenario for the year 2001. The methods implemented for the baseline scenario will be a useful instrument to evaluate the changes that are currently occurring and to respond to policy questions arising from future developments.

We have divided the process into the following different key elements according to the full chain approach illustrated in the graph (Figure A2).

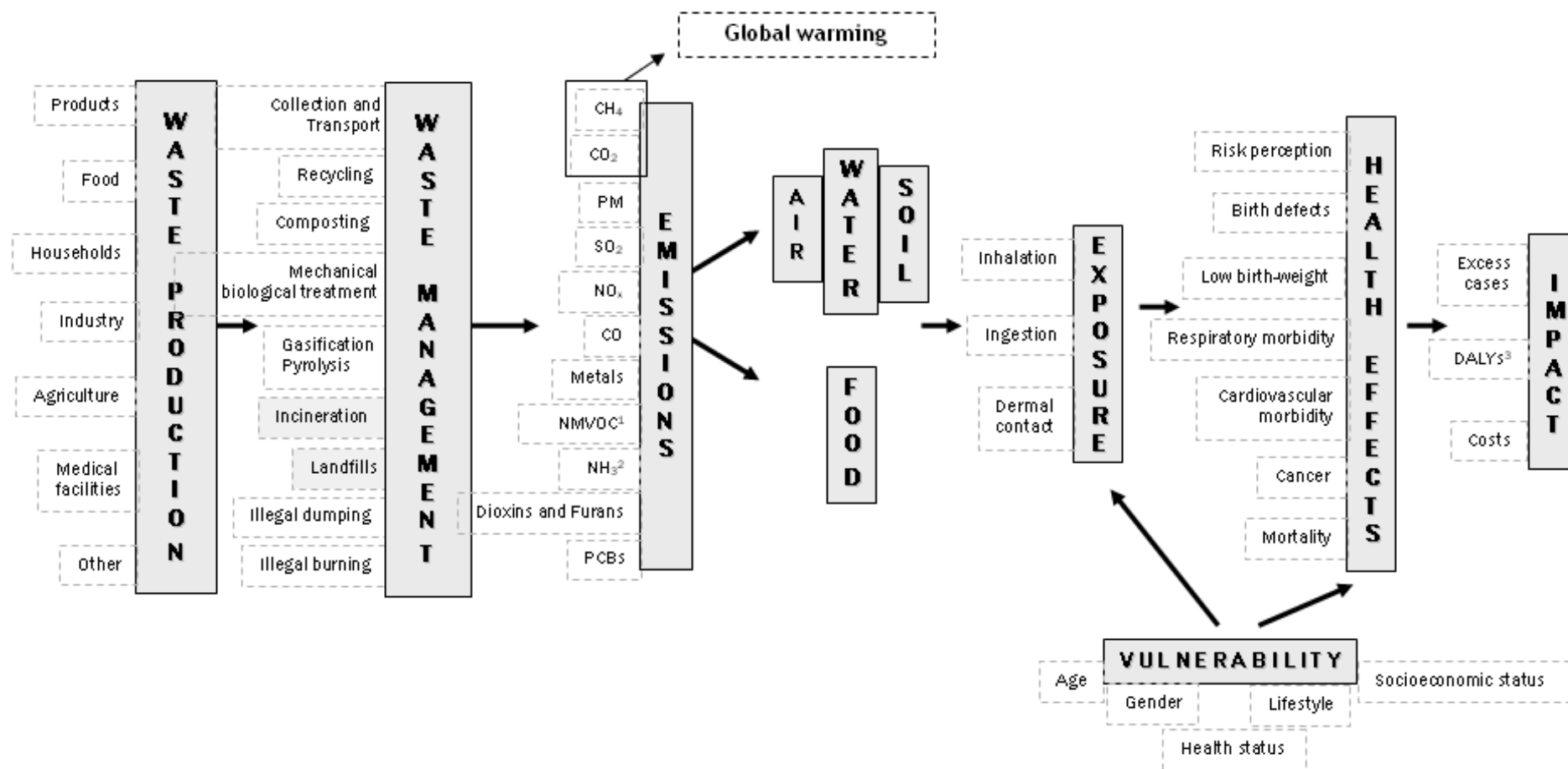
- From generation to management of waste. Describe waste generation and waste management policies for each country.
- From waste management to emissions of pollutants. Given the baseline scenario, evaluate emission data for the main waste technologies and estimate total emissions of air pollutants at the country level.

- From emissions to population exposures. Using geographical information systems (GIS) and dispersion modelling, 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.

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

- There are substantial environmental effects associated with waste transport for both recycling and disposal and we will try to consider transportation in the evaluation of total emissions.
- The quantification of illegal practices of dumping and burning is extremely difficult and only a qualitative assessment will be performed.
- The emission factors that we will consider are based on facilities under normal operational circumstances. There is the possibility of accidental releases that should be considered but are difficult to be quantified.
- Although all major waste management activities will be considered, the focus for the additional steps will be based on incinerators and landfills representing the main ways of waste disposal in the baseline scenario.
- Although pollutants from waste disposal practices are released into all environments, only emissions into ambient air will be taken into consideration in the full assessment, due to the lack of data on emissions into soil and water.

Figure A2. The full chain approach: from waste production to health effects



Note: ¹Non-methane volatile organic compounds; ²Ammonia; ³Disability-adjusted life-years.

A9. Health impact assessment of waste management facilities

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A9.1 Introduction

Health impacts may be assessed by considering the hazard, the pathways to a receptor, and the likelihood of that receptor being adversely affected by the hazard (the “sources-pathway-receptor” methodology). Information that is required includes:

- the type and quantity of emissions from the waste;
- the location and size of populations that could be affected by those emissions; and
- how they could be exposed to the toxic substance and the impact on their health of this exposure.

The impact of exposure of toxic chemicals on health is only known for a very limited number of chemicals. Therefore the ability to carry out a quantitative HIA is severely constrained by the very limited scientific evidence that is available. The risks to health are assessed using available or estimated emissions data, pathway modelling, and human dose estimates. The dose estimates are then related to standards or risk estimates based on prior research. In general it is helpful to quantify impacts on health and where the data are insufficient the best assessment should be made accepting that some degree of uncertainty will be unavoidable. Different methods are used to assess the health impacts of existing waste management facilities. One approach involves measurement of self-reported illness but these data suffer from participation and recall bias. Another approach compares disease incidence between exposed and comparison populations, either making use of existing disease registries (for example cancer or congenital malformations) or special surveillance systems. Increasingly, biological tests, which measure organ damage or dysfunction, are employed in exposed and control populations.

A9.2 Landfills

Each landfill site is unique with respect to age, quantity and type of waste contained, local meteorology, hydrogeology, and engineering control of leachate and landfill gas (LFG). The United States Agency for Toxic Substance and Disease Registry (ATSDR) have characterized emissions from hazardous waste sites (Amler & Lybarger, 1993, De Rosa, Stevens & Johnson, 1998, Johnson, 1997) and are studying the hundred or so trace chemicals which may be measured in leachate or LFG at such sites (De Rosa et al., 1996). People in the vicinity of hazardous waste sites may also be exposed to chemical mixtures (Etkina & Etkina, 1995, Hansen et al., 1998). Airborne exposure may lead to people inhaling constituents of LFG or emissions from LFG flares or particulate matter. It is in general unlikely that any abstraction points for public drinking-water supplies will be near the site but leachate contamination of nearby private water supplies is a possibility. If offsite soil is contaminated by atmospheric deposition or surface water then exposure may also occur by skin contact, ingestion of soil by casual hand to mouth contact, or by eating crops grown in the soil. Exposure of children is of particular relevance since young children are more likely to inadvertently ingest dust adhering to their hands. In the United States, results of public health assessments conducted at 167 waste sites during 1993-1995 identified complete or potentially complete exposure pathways involving 56 substances at 10% or more of the sites. Of these substances, 19 are either known or anticipated human carcinogens and 9 are associated with reproductive or endocrine-disrupting effects.

The ATSDR (1992) has also produced a list of seven priority health conditions, which might possibly be caused by exposure from waste sites. These are birth defects and reproductive disorders, cancers (selected sites), immune function disorders, kidney dysfunction, liver dysfunction, lung and respiratory diseases, and neurotoxic disorders. The particular vulnerability of children and pregnant women is a consideration. Most multi-site studies have concentrated upon congenital malformations but increased bladder cancers and leukaemia have been reported in women residing in areas likely to be exposed to landfill gas (Lewis-Michl et al., 1998). There have also been a large number of health surveys, which have relied upon residents reporting symptoms through questionnaires (Vrijheid, 2000). These may be subject to reporting bias but nevertheless indicate that concerns can have impacts on health.

A9.3 Incinerators

The substances of principal concern with regard to emissions from waste incineration are set out in the Waste Incineration Directive. Dioxins and furans, polycyclic aromatic hydrocarbons, and heavy metals (arsenic, nickel, cadmium, and chromium) are perhaps of greatest concern. The majority of published studies concentrate on the effects of exposure to emissions from the older generation of incinerators, which were phased out in the United

Kingdom after the introduction of stricter emission controls implemented through the Integrated Pollution Control (IPC) regime. In the United Kingdom, a large epidemiological study by Elliot and colleagues of the Small Area Health Statistics Unit (SAHSU) examined an aggregate population of 14 million people living within 7.5 km of 72 municipal solid waste incinerators. As a result, the Department of Health's Committee on Carcinogenicity (2000) concluded that "any potential risk of cancer due to residency (for periods in excess of ten years) near to municipal solid waste incinerators was exceedingly low and probably not measurable by the most modern techniques". Several studies have examined possible adverse effects on respiratory health among people living near incinerators (Hu, Hazucha & Shy, 2001) and failed to show any excess of acute chronic respiratory symptoms.

A9.4 Composting

Composting is a complex aerobic microbiological process by which the organic fraction of municipal solid waste and other organic wastes are converted into compost products. Composting organic materials produces biological aerosols (bioaerosols) consisting of actinomycetes, bacteria, fungi, protozoa, and organic constituents of microbial and plant origin (Fischer et al., 1999, Gilbert & Ward, 1998, Millner et al., 1994, Van der Werf, 1996). The Environment Agency (EA) reported (EA, 2001) a monitoring programme of key environmental emissions from various types of composting facilities treating various types of compostable waste. On many occasions the concentrations of bioaerosol measured both upwind and downwind of the sites exceeded 1000 cfu/m³ bacteria, 300 cfu/m³ gram-negative bacteria, and 1000 cfu/m³ fungi.

Bioaerosols produced by composting have the potential to produce adverse health effects such as aspergillosis, hypersensitivity pneumonitis and exacerbation of asthma (Browne et al., 2001, Bunker et al., 2000, Douwes et al., 2000, Epstein, 2001). There is also potential for disease if pathogens survive the composting process and are present in bioaerosols. Faecal contamination of raw material is highest when it incorporates large quantities of urban wastewater sludge or farm wastes. However, household wastes may contain human and domestic animal faeces and municipal solid wastes consist of about 1% by weight disposable nappies of which about one-third are soiled with faeces. Such faecal material may be contaminated with potentially pathogenic bacteria (for example Salmonellae), protozoa (for example Cryptosporidium parvum, Giardia lamblia), worms (for example Toxocara spp), or enteric viruses (Hepatitis A, Poliovirus, Coxsackie) (Pahren & Clark, 1987).

The risk to health for an individual exposed to bioaerosol from composting operations depends upon the concentrations in air of different components of the bioaerosol as well as personal exposure and prior health status. However occupational health and individual case reports demonstrate the potential for health risk in uncontrolled settings. *Aspergillus fumigatus* is an opportunistic pathogen in that it colonizes and infects individuals who are immuno-compromised. Hypersensitivity pneumonitis (extrinsic allergic alveolitis) may result from repeated inhalation of and sensitisation to a wide variety of organic aerosols including bacteria and fungi. It is completely reversible if antigen exposure ceases but continued exposure commonly leads to progressive interstitial fibrosis (Rose, C. S., 2000). Inhalation of specific allergens is a well-recognised cause of exacerbations of asthma. Asthma may be caused by allergens of microbial or plant origin but the amounts of airborne allergens that sensitize and incite asthmatic or allergic episodes cannot be defined given the wide variation in host sensitivity.

A9.5 Waste collection–transfer–recycling

In future waste collection authorities will be under pressure to meet recycling targets which may necessitate fortnightly collection. Putrescent matter held in unsuitable domestic situations for too long may encourage infestation. Most published research on waste transfer or sorting sites has focused on occupational health and internal air quality. A wide-ranging review, undertaken in Denmark (Poulsen et al., 1995), reported respiratory symptoms, gastrointestinal symptoms and irritation of the eyes and skin in those exposed. A Finnish study (Kiviranta et al., 1999) highlighted that concentrations of micro-organisms and VOCs around a waste transfer site were higher than around landfill sites but a Canadian study (Lavoie & Guertin, 2001) concluded that microbial air quality outdoors 100 metres downwind was not affected by operations. Road traffic generated by waste movements has potential health impacts and includes the use of private vehicles using recycling/civic amenity sites. Many significant air pollutants are primarily generated by road vehicles (for examples PM₁₀, benzene, and NO₂).

A9.6 Anxiety and distress

HIA must take account of the health effects arising from public anxiety about health impacts of waste management facilities (be they actual or perceived). Several studies have reported data on psychiatric symptoms amongst residents living close to a waste disposal site. Only five of these studies included samples of unexposed residents as a comparison group. There was some evidence to support the hypothesis that residents exposed to hazardous waste facilities exhibit greater levels of psychiatric morbidity than residents who are not exposed to such sites. However, it seems likely that at least some of this association might be explained by response bias, measurement bias and confounding. Local incinerators appear to generate the greatest public concern but people also display anxiety about living close to landfills incorporating hazardous waste. Psychiatric disorder is common, disabling and burdensome

and any excess associated with waste disposal needs to be accurately quantified. Psychiatric morbidity amongst residents living close to hazardous waste sites might be improved through transparent and accurate communication of the health risks involved, with the aim of alleviating the heightened yet understandable concern in the exposed population. A well-run HIA process will do this at the same time as making more quantitative analyses of health risks.

A9.7 The way forward

The review of the different waste management options demonstrates that all produce emissions that have the potential to harm health. It is impossible to say that a strategy maximizing recycling and composting and minimizing incineration and landfill will reduce local health impacts. The areas where better evidence to support HIA of waste strategies most immediately needed are:

- more sophisticated spatial epidemiology of health outcomes married to dispersion modelling of emissions;
and
- more investigation of the role of confounding factors in determining psychological morbidity of individuals living close to waste facilities and evaluation of interventions directed to preventing psychological morbidity.

Local impacts on health from proposed waste management facilities must be considered by the local decision-making processes. HIA can contribute to these decision-making processes and may be submitted as evidence to the statutory authorities. The Chemical Hazard Management and Research Centre, University of Birmingham, has identified the principles that should underpin local health authority input and suggests key components of a public health assessment for IPPC applications (Chemical Hazard Management and Research Centre, 2001).

A10. Priority needs in research

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A10.1 Introduction

As far as the emission-to-health characterization is concerned, the recently suggested *environmental public health continuum* seems a very useful reference model (Albertini et al., 2006). The process *source/stressor formation-transport/transformation-exposure-dose/early biological effect-altered structure/function-disease* is adequate for application in waste treatment and in management settings.

Different types of environmental and health hazard are present in each step of the waste cycle: production (domestic, industrial)-separate collection (different strategies)-recovering/recycling of primary raw materials (for example, glass, paper, metals, drugs, batteries, etc.)(recycling industry)-pre-selection (by hand or machine) and recovering (industry or disposal)-composting (agriculture)-disposal, plus transportation.

Among the information needed for the environmental characterization, data about the waste site (type of waste, plant, landfill, management, etc.) and data concerning the area where the site is located (geomorphology, chemical composition of soil, surface- and ground-water, etc) must be known. Furthermore, information on pollutants emitted or released having impact on environment and health is necessary, as well as air emission and dispersion (measuring and modelling) and soil and water contamination (persistence, chemical degradation, biodegradation, bioavailability).

Usually, routine information that allows a useful environmental characterization is unavailable. In such a circumstance, systematic hazard scoring systems have been developed in the framework of large site assessment programs (Johnson, 1999) or public health assessment programs (see, as examples, United Kingdom DEFRA (<http://www.defra.gov.uk>) and New York State Department of Health (<http://www.health.state.ny.us>)).

In Europe, a hazard ranking of hazardous waste landfill sites has been proposed by the EUROHAZCON study on congenital malformations (Vrijheid et al., 2002b).

As far as the environment-to-exposure characterization is concerned, the availability of valid data on pollutants pathways is critical: food chain contamination (bioaccumulation, biodegradation, biomagnification) and routes of exposure (inhalation, ingestion, contact) are often totally or partially unknown. Progress has been made in measuring and estimating individual exposure, both by external monitoring devices (for example, passive dosimeters) and direct measurements of intake dose using biomarkers of exposure. However, the fact that “without exposure pathway information, it is difficult to relate biomonitoring results to sources and routes of exposure and develop effective health risk management strategies” must be taken into consideration (Albertini et al., 2006).

In exposure characterization the main issues are timing, latency and low doses. The definition of timing and latency requires a some knowledge of exposure and the etiology of the selected health outcomes. Low-level exposure effects are difficult to detect, particularly when rare endpoints in small populations are investigated (low statistical power), nevertheless some important successes in the field must be taken into consideration (Vineis, 2007).

In a hierarchy of exposure assessment, individual direct measurements are at the highest level, and community measurements and estimates at the lowest level. Intermediate levels, from highest to lowest, are: quantified area or ambient measurements in the vicinity of the residence or other sites of activity, quantified surrogates of exposure (for example estimates of drinking water use), distance from site and duration of residence, distance or duration of residence, residence or employment in geographic area in reasonable proximity to site where exposure can be assumed. The technological development of active and passive dosimeters, as well as the methodological activity to compare dosimetry and biomonitoring results, have provided useful insights (Baker, D. S. R., McCallum & Driver, 2001, Ross et al., 2007).

When direct individual measurements are not possible or not feasible, indirect estimation of exposure is used. These measures of exposure are as good as they closely approximate the individual exposure: the distance of the individuals' residence from the point source of pollution is the choice of several epidemiological studies, however some other methods are available or under development, for example those based on emission and diffusion models associated with georeferenced individual subjects (see Annex A15). These systems can support different designs of epidemiological studies (case–controls, case–crossover, etc.).

Direct measurements of exposure by monitoring of biomarkers of exposure are increasingly common, and, as they are able to give information of daily intake, absorption and body burden at the individual level, can be highly

informative. In association with environmental monitoring information they can characterize the emission–exposure link.

At the opposite end of the chain, the outcome characterization is equally important. Among the different components of disease definition – operational definition and diagnostic criteria, etiopathogenesis, particularly concerning the role of the environment, availability of data from routinely collected statistics or from ad hoc surveys, classification and coding, the latter assumes a high relevance when disease categories are established. Multidisciplinary work is needed to enable the inclusion of the main variables involved, both of statistical type (disease frequency, size of the study population, estimable risk to detect), and of medical type (etiopathogenesis, clinical aspects, diagnosis ability). In general, it is preferable to select as endpoints specific diseases for which an association to waste exposure is recognized or suspected, for example, a specific cancer as NHL or a specific congenital malformation as cleft palate, but finer classification can lead to low study power. On the other hand, selecting a group of diseases (for example, all cancers or all oral clefts) the power as well as the etiological variability within the group increases, with a consequent more difficult interpretation.

A10.2 Emission-Exposure-Health effects characterization

Following the fast developments of the last decades, methods and tools are today available to design and conduct advanced epidemiological studies able to deal with complex situations. In the evolution of epidemiology, genetics, molecular genetics and genomics assume high relevance. Advancement towards a better integration of epidemiology with genetics aims at improving the understanding of diseases etiology at a population level. The development of methods for large, population based surveys and for the joint investigation of environmental and genetic hypotheses represents a good opportunity in the field of health and waste management (Palmer, L. J., 2004).

Using GIS and/or global positioning system (GPS) tools, detailed geographic information is obtainable, for example georeferentiation of point, linear or diffused sources of pollution. The possibilities of application in epidemiology are manifold and probably underestimated.

Available statistical methods are adequate to deal with many problems such as: multiple testing, power/sample size, multilevel analysis, control confounding, interaction (for example due to socioeconomic status and other determinants), selection of reference groups or population, temporal, spatial, time-space interaction, clustering, cluster detection analysis, random and spatially structured variability.

A10.3 Focusing on human biomonitoring

To realize the potential of HBM, investment in research is needed to address the critical knowledge gaps that hinder the ability to use HBM data and interpret what they mean with respect to risks to public health (National Research Council, 2006). The examples reported in Table A5, plus the recent French experience as presented by Viel (see Annex A16), clearly show the complexity involved in performing and interpreting studies on workers and residents. Issues like transferability of results from workers to general population, difference of strength of evidence between “classical” epidemiology and HBM based studies, implications for communication and decision-making, are among the research priorities.

Four recommendations are as follows.

1. There seems to be a lack of consistent strategies for inclusion/exclusion criteria of chemicals in HBM studies. Moreover, susceptible and vulnerable subpopulations (infants and children, particularly exposed groups, etc.) are generally not included in large-scale surveys due to difficulties in their identification and sample collection. A strategy for HBM development and population HBM based on population exposure and public health concerns appears thus desirable. A selection of chemicals based on their potential to produce health effects may be the key criteria to develop this strategy, and the aspects to be considered are:
 - a. data on, or evidence of, exposure of population or subpopulations;
 - b. toxicology data associated with health effects;
 - c. HBM and exposure information concerning susceptible population groups; and
 - d. environmental pathways, especially to estimate the potentiality of persistence.

The United States experience, where several federal agencies are collaborating to develop and apply this kind of strategy, represents the reference example.

2. The technical ability to detect chemicals has outpaced the ability to interpret health risks, and predictive knowledge of impact of technology lags behind technical knowledge (techne), (Jonas, 1979).

Epidemiological and exposure assessment studies have not sufficiently and adequately incorporated HBM for evaluating individual, community and population health risks; therefore HBM based epidemiological and exposure assessment studies and surveillance programs must be developed. This can also lead to significant progress in investigating health effects due to low-level environmental chemicals exposure. Exposure analysis for subgroups of the population needs detailed data of the environmental matrices state to evaluate chemical exposure pathways. In the toxicological field particular attention should be paid to experimental studies based on HBM, so as to assess dose–response relationships. Pharmacokinetic models are fundamental in understanding HBM response, since they are able to assess the influence of factors like metabolism as well as sampling time.

3. Communication strategies are often absent or poor, especially a specific design of communication process, analysis of information needed and tools to be incorporated in the design, implementation and appraisal of HBM studies. The development of communication methods and tools for developing HBM studies, as well as for reporting the results of the studies at the individual, community and population level, assumes a high priority.
4. Ethical aspects concerning informed consent and interpretations of results need to be dealt with with care. Many issues are very difficult to manage, mainly because they are strictly connected to each other. For example, anonymous samples and limitation of communication and follow up, or general information versus detailed information and the corresponding problems of feasibility and privacy. Analysis of bioethical issues and the development of ad hoc methods to inform subjects about both the general study and the specific HBM purposes are recommended. In general, to allow HBM use in large scale epidemiological studies, a specific approach would be advisable, particularly to improve laboratory methods in terms of quality of data, analytic sensitivity, easier and cheaper analytic methods (for breast milk, exhaled breath, saliva, etc.).

Table A5. Complexity in performing studies on workers and residents and in their interpretation

TITLE	STUDY SUBJECTS	MAIN RESULTS	AUTHORS
Negative association between serum dioxin level and oxidative DNA damage markers in municipal waste incinerator workers.	Workers Municipal waste incinerator	Lymphocytic 8-hydroxy-2'-deoxy-guanosine (8-OH-dG) level showed a negative association with serum dioxin level. Urinary 8-OH-dG level correlated positively with smoking index; serum dioxin level did not lead to an increase of urinary 8-OH-dG by oxidative DNA damage, but up-regulation of the primary defenses with oxidative damage and/or the DNA repair system might occur. The smoking habit may be a more important factor of oxidative damage than dioxin. Total TEQ value of the subjects was almost the same of the general population.	(Yoshida et al., 2006)
Increased cytochrome P4501B1 gene expression in peripheral leukocytes of municipal waste incinerator workers	Workers Municipal waste incinerator	CYP1B1*3 (a new gene sub-family of cytochrome P450) polymorphism seemed to modify the effects of exposure on CYP1B1 gene expression. It is possible that occupational and environmental PAHs/dioxins exposures are different in the magnitude, chemical structures and frequencies of exposure. The present and other studies suggested that elevated CYP1B1 gene expression was associated with occupational, but not environmental, exposure to PAH/dioxins. The discrepancies in findings from studies of occupationally- and environmentally-exposed subjects merit further clarification	(Hu et al., 2006)
Comparisons of levels of polychlorinated dibenzo-p-dioxins/dibenzofurans in the surrounding environment and workplace of two municipal solid waste incinerators.	Areas outside and inside incinerators plants	The total PCDD/F concentrations in the workplace air were 5–13 times higher than that in the outdoor air; the PCDD/Fs WHO-TEQ concentrations in the workplace air were 5–15 higher than that in the outdoor air. In addition, the individual congener concentrations also exhibit a similar phenomenon. Results reveal that workplace environments are much worse than outdoors in two MSWIs	(Shih et al., 2006)

A10.4 Role of epidemiology in waste and health field

The social context plays a fundamental role in planning and performing studies on waste risk to health. Risk perception, based on local knowledge and experience, determines the attitude of the general population and influences viewpoints of different stakeholders. For example, limitations and biases of epidemiological studies may be ignored or dismissed by those backing a causal association, while they can be used by others to diminish findings.

Studies on waste-related risks normally do not address the methodological question of whether risk factor (so called “black box”) epidemiology or eco-epidemiology (so called “Chines box epidemiology”) is more appropriate. To evaluate available studies in this context, the debate on risk factor epidemiology and new paradigms may be useful (Susser, M. & Susser, 1996a, b). These two different schools of thought provide important material for reflection.

Identification of risk factors is vital for public health research. By the mid-1980s, a large body of valuable study design options and analytical methods was developed to investigate causal relationships and to identify implications for prevention, even in the absence of an understanding of the biological processes that linked exposure to disease, and of the social context that gave rise to them (Savitz, 1994). Black box epidemiology supporters argue that restricting epidemiology to the identification of risk factors needlessly restricts the scope of the discipline. Current developments call epidemiologists to adopting a wider and richer framework that subsumes but does not discard black box epidemiology (Susser, E., 2004). On the other hand, the accumulation of evidence on initially unexplained risk factors can ultimately provide the basis for a deeper understanding of disease etiology, and help develop sound public health preventive action (Greenland, Gago-Dominguez & Castelao, 2004).

Recently, four lines of work have been suggested for strengthening epidemiology (Susser, E., 2004).

1. Address “multilevel” causation (society at macro level, behaviour at individual level, cells/genes at micro level); disease afflicts a person but since its prevention regards each level, different approaches are needed to identify the causal processes at each level (Rose, G., 1992, Schwartz & Diez-Roux, 2001).
2. Epidemiological designs must adapt to incorporate and exploit gene biology information and address gene–environment interactions, to investigate jointly genetic and non-genetic risk factors.
3. Life-course approaches, from early life experiences to health and illness over life should be studied with specific methodology (Kuh & Ben-Schlomo, 2004).
4. Contrary to fragmentation into sub-specialities a general broader and more unified framework should be developed that can sustain the coherence of the discipline.

An integrated approach that assumes such four lines, as proposed in eco-epidemiology, seems adequate also in waste and health. Integrating risk factor epidemiology within a broader framework has been previously discussed by Krieger (1994) and McMichael (1999); a recent paper develops these ideas for a transition from the traditional paradigm of risk factor epidemiology to eco-epidemiology: “the risk factor storm and the shoring of ecology” (March & Susser, 2006). In waste and health, as in many other fields, a risk factor and an eco-epidemiology could integrate each other. Considering the relevant public health and general value of waste in the developed and developing countries, it is recommendable to design studies based on HBM exposure assessment and develop guidelines also to interpret study results.

A11. Monetary valuation of impacts and cost-benefit analysis*

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*based on article to be published in *Waste Management & Research*

The methodology for evaluating the impacts and damage costs (“external costs”) due to pollution from waste treatment is described and results are presented, based on the Externalities of Energy (ExternE) project series of the European Commission.

The damage costs of landfill and incineration of municipal solid waste are compared, with due account for energy and materials recovery, as well as possible differences in transport distance. We have not been able to quantify the total damage costs of leachates because of the complexity of the environmental pathways and of the long-time horizon of some persistent pollutants, but we consider an extreme scenario to show that they are not worth worrying about in the sense that reducing the pollutants in leachates beyond current regulations would bring negligible benefit compared to the abatement of other sources of the same pollutants.

The damage costs due to the construction of the waste treatment facility are negligible. The damage costs of waste transport, illustrated with an arbitrary choice of 100 km roundtrip by a 16 tonne truck, are also negligible. The benefits of materials recovery make a small contribution to the total damage cost. 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€/t_{waste}, extremely dependent on the assumed scenario for energy recovery. For landfill the cost ranges from about 10 to 13€/t_{waste}; it is dominated by greenhouse gas emissions because only a fraction of the CH₄ can be captured (here assumed to be 70%). Amenity costs (odour, visual impact, noise) are highly site-specific and we only cite results from a literature survey which indicates that such costs could make a significant contribution, on the order of one euro per tonne waste.

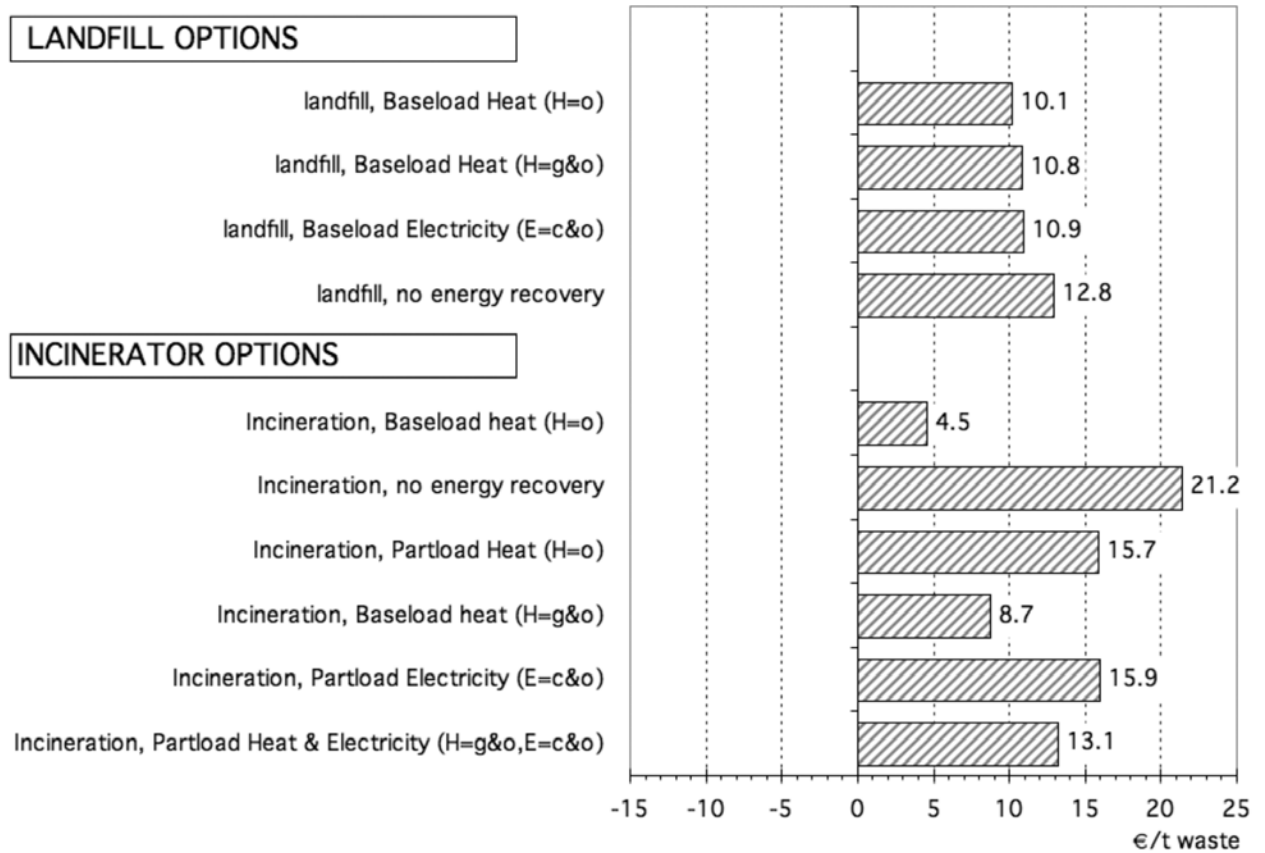
A summary of the total damage cost for all the waste treatment options is shown in Figure A3. The total cost of incinerator emissions, without energy recovery or materials recovery, is 22.9€/t_{waste}. Most of that is due to particulate matter, NO_x, SO₂ and CO₂. Toxic metals and dioxins contribute only 2.5€/t_{waste}, mostly because of Hg and Pb. The contribution of dioxins is negligible, only 0.1€/t_{waste}, thanks to the low emission limit of the Directive EC of 2000.

Unlike incinerators, the damage cost of landfill does not vary as much with type of energy recovery because in any case the amount recovered is relatively small. The benefits of energy recovery are largest if the heat can be used directly for process heat or district heating systems with sufficiently large constant load. Electricity production brings far lower benefits than heat because of the poor conversion efficiency of incinerator heat (compared to central station power plants).

The results presented in this paper are for typical conditions in France, but they can be adapted to other sites and other countries if the respective damage costs per emitted pollutant are known. Even without carrying out new calculations using the EcoSense software of the ExternE project series, one can estimate the damage costs per emitted pollutant using the “uniform world model” described in the full paper and in several publications of Spadaro and Rabl (for further details see <http://www.arirabl.org/>).

The uncertainties are large and they have different effects on different policy choices. Comparisons between landfill and incineration are especially sensitive to the uncertainty of greenhouse gases because they play such a large role for landfills. Comparisons between different types of energy recovery for incinerators, on the other hand, also depend on the other pollutants. The emission of greenhouse gases per tonne of waste depends strongly on the type of sorting and pre-treatment that is carried out. For example, if paper, cardboard and plastics are removed, the biological fraction of the remaining waste increases and so do the CH₄ emissions from landfill.

Figure A3. Results for total damage cost for all options. If electricity displaces nuclear, damage costs are essentially the same as for the case without energy recovery. Amenity costs are not included; they are very site-specific and could make a contribution on the order of one euro/t_{waste}



In decisions about waste treatment the full social cost (the sum of private costs and external costs) should be taken into account. According to data we have seen, the private costs of landfilling are less than half of those of incineration: very roughly around 40€/t_{waste} for landfilling and around 100€/t_{waste} for incineration. However, we emphasize that costs (private costs plus damage costs) are not the only criterion for choosing a treatment option for MSW. There are almost always additional criteria, especially the preferences of the local population, that may be difficult or impossible to express in monetary terms. Land use and land availability are crucial. In many regions of Europe land is so limited that incineration is the preferred choice even if its cost is higher. Such non-monetary criteria can be taken into account by means of a multicriteria analysis, preferably in consultation with the stakeholders.

A12. Case studies: an introduction

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The rationale for having a fairly extended session on case studies may be summarized as follows. The review of available evidence leaves many open questions and new scientific studies may fill gaps in etiologic knowledge; still, is not possible to straightforwardly foresee how research findings will be used in decision-making.

In order to ensure that epidemiological studies may provide a real contribution to public health development in the domain of waste management, it is in first place necessary to bring to light the underlying assumptions of study design, namely the background causal evaluation being adopted, the specification of ascertained or suspected causal agents of disease being investigated, their corresponding levels of definition and the specification of estimated exposure patterns.

Special emphasis should be addressed to both validity and feasibility aspects, namely the use of current health statistics versus ad hoc surveys, choice of exposure metrics (see above) and level of spatial aggregation, the choice of methods for taking into account potential role of socioeconomic deprivation and the insight in pathogenetic mechanisms.

In order to define a shared frame for causality assessment, there is the need to discuss the applicability of traditional causality criteria to complex exposure mixtures involved in waste cycle, the issue of consistency versus biological plausibility and the role of causal judgment in preventive action and in precautionary policies.

From these premises, it may be possible to reach a common ground for expert advice, communication and community empowerment. It may in particular be advisable to define health-based criteria for setting priorities for environmental remediation and the contribution to health impact assessment procedures of new landfills and incinerators.

Finally, this process may contribute to correctly design information of the general public and promotion of participatory activities in affected communities, in order to foster their autonomy.

A13. Landfills, case study (Denmark): pharmaceutical and other chemical waste in the dunes. Denmark's largest chemical landfill in Kaergaard, Denmark

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During a period of almost 20 years, the dunes in Kaergaard Plantation on the west coast of Jutland were used as a deposit for waste from a large pharmaceutical industry in the nearby town of Grindsted. Few years after the depositing started, the neighbours started to complain about the smell and shortly after, it became clear that the waste began to appear on the beach. In 2007, the deposit is still there but plans have been made to clean it up. The technical problems are large but the question on how to finance the clean up is even more difficult to solve.

A13.1 History

A small chemical factory was established in the town of Grindsted in western Jutland in the 1920s. It gave new jobs in a remote region and created prosperity in an area dominated by agriculture. After the World War II the factory focused on the production of medical products including barbiturates, lithium, sulphonamides and vitamins. Mercury, chlorinated hydrocarbons and other chemicals were a part of the production. An increasing amount of waste was produced. Solid waste was deposited in a 500 meter long deposit along a railway while the waste water was led to a nearby river. After some years, the discharge of waste water gave many complaints. The town water supply became polluted and after some time, the local authorities ordered the company to stop the discharge. The local authorities assigned a place in a remote plantation near the west coast of Jutland where the company could discharge the fluid waste. During the period from 1956 to 1974, a total of more than 280 000 tonnes of chemical waste was deposited in five different pits some 500 metres from the coastline.

A13.2 Effects on the surroundings

The deposition of waste in the dunes soon gave problems. Smell from the pits gave complaints from neighbours living several kilometres from the deposit. As a result, the pits were covered with roofs which diminished the problem. In 1962, a foul smelling foam appeared on the beach and the local medical officer demanded the establishment of a ban against bathing on a 700 meter long stretch on the beach. Some groundwater wells inland seemed unaffected. Due to increasing concern about the long-term effects of the deposition of waste, they were finally stopped in 1974. The company instead got a permission to dump the waste in the North Sea and the Atlantic but local and international pressure stopped that practice within a few years.

A13.3 The deposit today

After stopping the deposition of waste, the deposit was left alone for decades. The ban on bathing was continued but no other initiatives were taken. In the beginning of the 1980's, a growing interest in pollution and the environment in general gave rise to some initiatives. The extent of the pollution was determined and some analyses performed. It was decided to leave it as it was and follow the situation. During the next 25 years, numerous investigations and analyses were performed. A considerable degradation takes place but the exact speed of the degradation is unknown and new substances are created, including the highly toxic vinylchloride. No chemicals which are known to bioaccumulate seem to be present although traces of chlorophenols have been detected in one pit. The quantities are small and are likely to be degraded before they reach the coast.

The coastal waters are polluted and the maximal limit for drinking-water quality is exceeded for several substances, including aniline, sulphonamides and vinylchloride. Bathing is therefore still prohibited. In the pit areas, air quality limits for volatile compounds like chlorinated hydrocarbons are exceeded and the same may be the case in some areas near the coast. Due to the windy weather in the area, the genuine exposure is likely to be lower than the calculated ambient air concentrations. Visitors to the area are advised not to settle in the area but passing through the area on foot is accepted. These simple restrictions on use are seen as sufficient to ensure the health and safety of visitors to the area. Nobody live permanently in the vicinity of the landfill and there are no evidence of any exposure in the nearest exposed settlements. The effect on the maritime environment seems limited but is not studied in detail.

A13.4 The future of the landfill

A clean-up of the two largest "hot spots" will start during 2007. The aim is to decrease the total amount of chemical waste and thus reduce the flow of chemicals into the North Sea. It will take generations before the beach is suitable for bathing. The industrialisation and establishing of the factory in Grindsted will leave its tracks for centuries.

A14. Landfills, case study (Italy): epidemiological studies around the municipal waste landfill in Turin, Italy

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A14.1 Introduction

The municipal waste landfill of Turin is one of the biggest in Italy; it is set in the northwest part of the city and covers an area of 89 hm² (the name of the area is *Basse di Stura*). It has been operating since 1975 and has a capacity of 20 920 000 m³, serving 51 municipalities and 1 380 000 inhabitants. At the moment it collects 700 000 tons of urban solid waste per year with a relevant impact on environment and on air quality. Since 2001 the site has been declared of national interest for reclamation and decontamination. Its ultimate closure is expected for the year 2010.

The northwest part of the city is characterized by multiple industrial settlements and is substantially inhabited by underprivileged population, particularly disadvantaged under cultural and economic point of view. The deprivation index (Cadum et al., 1999), built in order to control for socioeconomic deprivation in epidemiological analyses, shows here the presence of the worst conditions in the city, along with a high prevalence of immigrants from the southern (poorest) part of the country.

Several epidemiological descriptive studies have been carried on during the last years in order to characterize the health profile of the resident population, trying to take into account several confounding factors, and in particular the socioeconomic confounding and for the correlated presence of other emissive sources.

A14.2 Point source mortality study. Methods and main results.

A first descriptive epidemiological study was carried on in 2002 and published in 2004 (Mitis et al., 2004). We analyzed mortality data at municipal level for the period 1990–1994 and at small area level for the period 1971–1999.

These two studies had different design and showed different results. The mortality study at municipal level was a descriptive geographical study based on current statistics of mortality. We calculated crude and standardized rates, cumulative risk, and standardized mortality ratios (SMR) (adjusted for deprivation index), separately for men and women.

The results showed significant excess risks for rectum and colon cancer (SMR = 109 for both sexes), for liver cancer (SMR = 126.4 for men and 105.3 for women), for bladder cancer among men (SMR = 116.8) and for lung cancer among women (SMR = 123.5).

The mortality study at small area level had a hybrid design. It is a modified cohort study, in which information at individual level is obtained through record linkage procedures between anagraphic, census and mortality data. Census data include information also about residence (street and civic number); SMR were then computed separately for gender, calendar period, and distance (within 3.5 km) from the centre of the waste landfill, controlling for education and area of birth.

Results showed significant decreasing risks according to distance for lung cancer among men ($p = 0.033$) and women ($p = 0.002$), for bladder cancer among men ($p = 0.04$), and leukaemia among children (both sexes together) ($p = 0.017$).

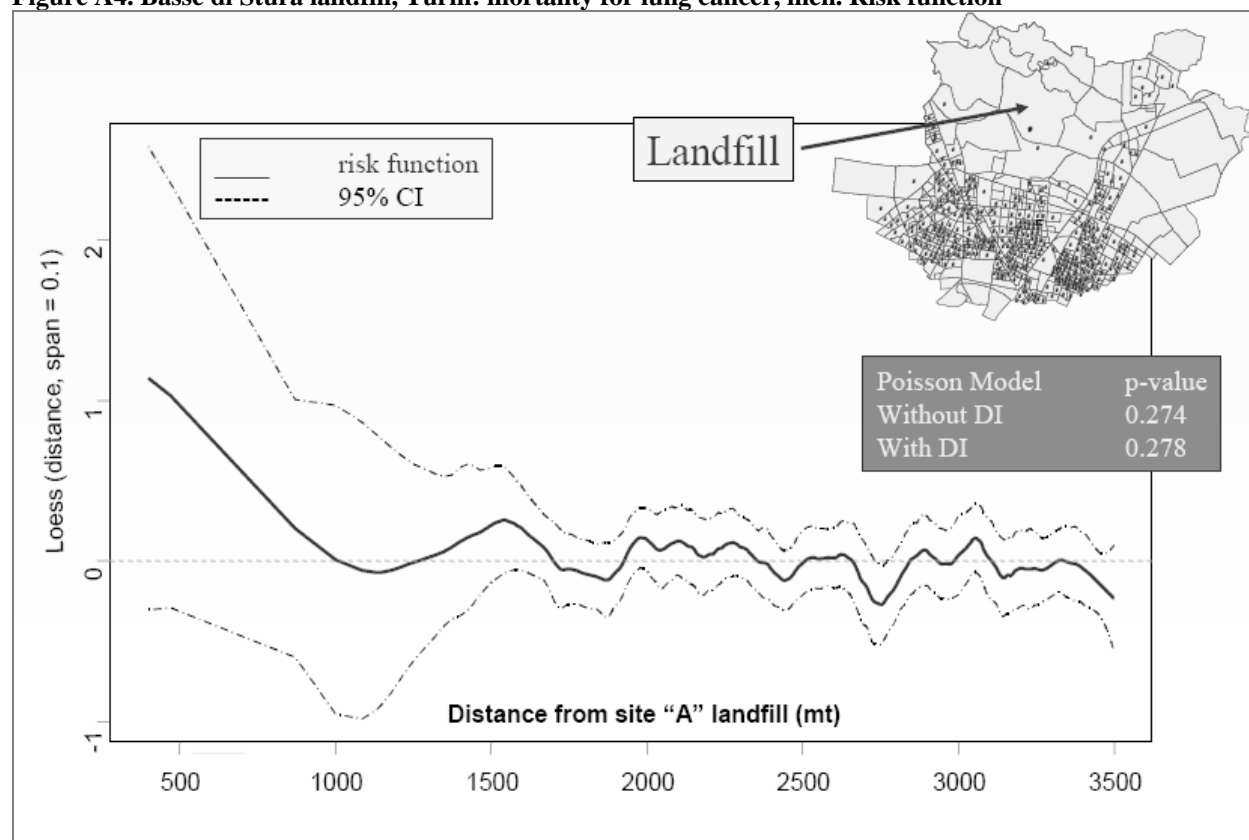
The results of the Stone test calculation for lung cancer among men and the related figure of the risk function are reported in Table A6 and Figure A4.

Table A6. Basse di Stura landfill, Turin: mortality for lung cancer, men. Stone test

BANDS (KM)	RR	SINGLE BAND					CUMULATIVE VALUES		
		OBS ¹	EXP ²	SMR	95% CI		OBS	EXP	SMR
0 – 0.5	3.611	2	0.554	3.611	0.047	10.058	2	0.554	3.611
0.5 – 1.0	1.111	0	0.106	0	-	-	2	0.66	3.032
1.0 – 1.5	1.111	17	15.203	1.118	0.602	1.709	19	15.863	1.198
1.5 – 2.0	1.103	255	231.17	1.103	0.973	1.274	274	247.033	1.109
2.0 – 2.5	1.079	383	355.112	1.079	0.977	1.194	657	602.145	1.091
2.5 – 3.0	1.018	390	387.062	1.008	0.915	1.115	1047	989.206	1.058
3.0 – 3.5	1.018	405	393.978	1.028	0.934	1.135	1452	1383.185	1.050

Note: ¹Observed values; ²Expected values
Source: adapted from Mitis F. et al., 2004

Figure A4. Basse di Stura landfill, Turin: mortality for lung cancer, men. Risk function



Source: Mitis F. et al., 2004

A14.3 ARPA microgeographical study. Methods and results

In 2003 the Epidemiologic Unit of ARPA Piedmont carried out a second descriptive microgeographical study around the landfill area for a distance of three kilometres, as a part of programmed studies foreseen during the decontamination activities (Ivaldi, Demaria & Cadum, 2003) (see Figure A5).

The study population corresponded to a cohort of 3100 individuals, residents within three kilometres from the centre of the landfill, followed up since 1980 (as part of the Turin Longitudinal Study (1971–2004)). Statistical analysis by census tract was carried out not only for mortality (1980–2002, 36 causes of death) but also for hospital admissions (1995–2002, 36 causes), for congenital malformations (1995–2002) and low birth weight (1995–2002). Crude and age standardized rates were computed, as well as SMRs and hierarchical Bayesian estimators (BMR) according to the Besag York and Mollié statistic (Besag, York & Mollié, 1991).

The main results for mortality (follow-up 1980–2002) showed excesses (not significant) for lung cancer (SMR = 119, 95% CI: 51–234), bladder cancer (SMR = 159, 95% CI: 64–327), leukaemia (SMR = 132, 95% CI: 57–260), endocrine pathologies (SMR = 139, 95% CI: 67–256), diabetes (SMR = 136.5, 95% CI: 50–297), ischaemic heart disease (SMR = 107, 95% CI: 76–146) among men. Among women significant excesses were

detected for endocrine pathologies (SMR = 196, 95% CI: 101–341)) and diabetes (SMR = 206, 95% CI: 100–380); not significant excesses were reported for lung cancer (SMR = 124, 95% CI: 46–271), stomach cancer (SMR = 157, 95% CI: 51–366), cirrhosis (SMR = 142, 95% CI: 52–308).

The analysis of hospital admissions (follow-up 1995–2002) showed significant excesses among men for general respiratory causes (SMR = 121, 95% CI: 105–137), multiple myeloma (SMR = 432, 95% CI: 173–889); and not significant excesses for lung cancer (SMR = 154, 95% CI: 100–228), stomach cancer (SMR = 169, 95% CI: 62–368), bladder cancer (SMR = 129, 95% CI: 78–202). Among women significant excesses were detected for breast cancer (SMR = 171, 95% CI: 124–230) and neuropsychiatric causes (SMR = 135, 95% CI: 106–169); not significant excesses were detected for ovarian cancer (SMR = 160, 95% CI: 59–347) and lung cancer (SMR = 198, 95% CI: 80–408).

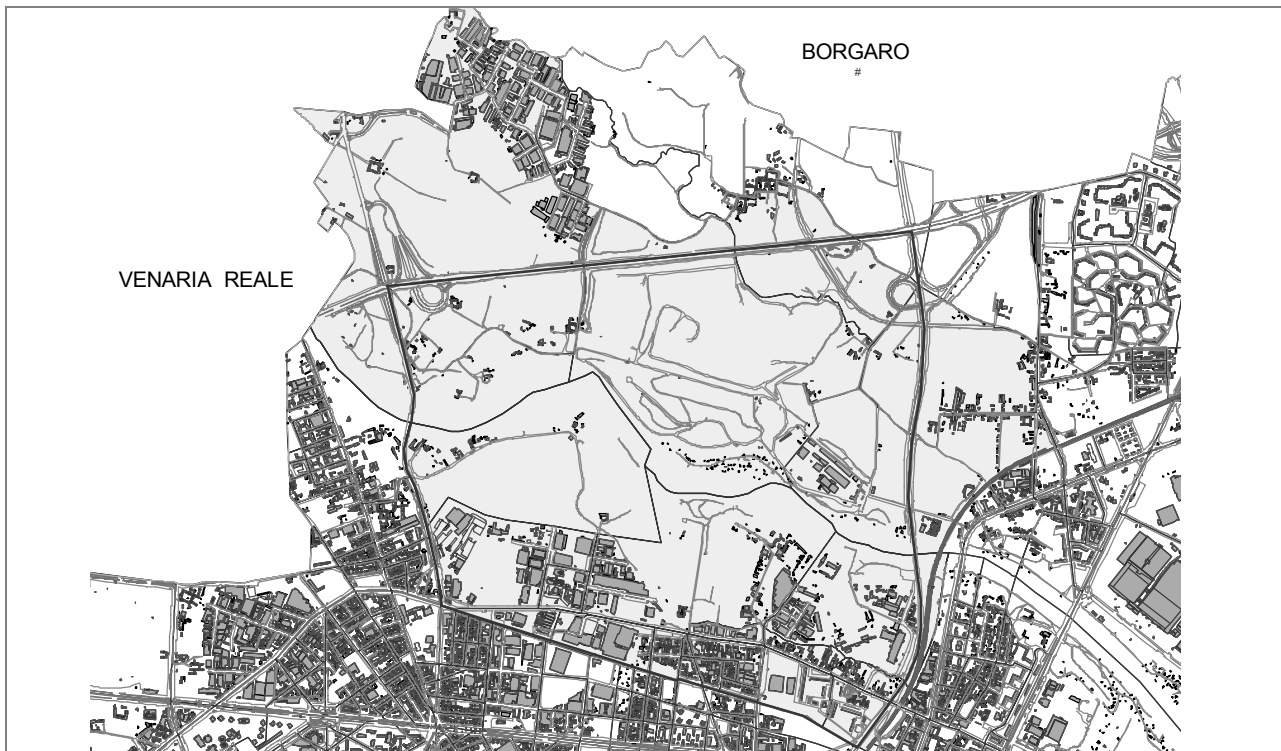
The analysis for congenital malformations (follow-up 1995–2002) were estimated from hospital admissions. No evidence of risk was detected. Fifteen cases for all kinds of congenital malformations were observed, with SMR = 95 (95% CI: 53.2–156.9).

Only four cases of low birth weight were noticed; this number is beyond any significative description.

A particular mention is due to the presence, in the study area, of a gypsy camp and to their health risk. High and significative respiratory disease risks were detected among the nomadic settlement, both for men and for women. A particularly and impressive high prevalence of almost all diseases were pointed out among children (but not for leukaemia).

The overall results (having seen in particular the pattern of causes of disease in excess) are consistent with a strong effect of a widespread socioeconomic deprivation present in the area; doubts remain anyway about possible effect of environmental exposures, mixed with social effect, particularly for the higher risks pointed out for all related respiratory causes.

Figure A5. Basse di Stura landfill, Turin: study area (in grey) of the ARPA microgeographical study

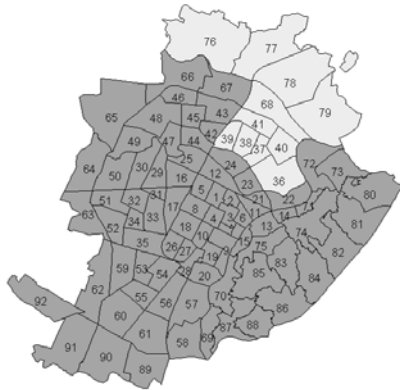


A14.4 The ARPA-ASL5 epidemiological study on respiratory diseases

In 2004 the Epidemiologic Unit of ARPA Piedmont carried out a third study in the area in collaboration with the Epidemiologic Service of ASL5, the Local Health Unit (Demaria et al., 2004). The study was requested by the sixth district of the city of Turin, in order to identify the causes and responsibilities of the excesses for respiratory causes in the area around the Basse di Stura landfill.

All the Sixth district (see Figure A6) showed in previous surveys higher risks of mortality for respiratory causes (International Classification of Disease, ninth revision (ICD IX) 460–519) and for lung cancer (ICD IX 162) in comparison with the rest of the city.

Figure A6. Sixth district, Turin city



The study was carried out at two levels. The first level corresponded to a geographical descriptive study of the whole district, while the second level corresponded to a multilevel analysis of geographical differences between inner areas in relation to environmental, socioeconomic and geographical variations.

The analyses were carried out for mortality and hospital admissions, for respiratory causes, for two age groups (15–64 and 65+) and for males and females, separately, controlling for age, educational level and birth area.

A separate study was additionally performed on all the cohort of occupied in waste treatment resident in Turin.

The hospital admission data for respiratory causes showed higher risks in the Sixth district in comparison with the rest of the city, in particular among the children (SMR = 201, 95%CI: 143–273). The mortality of the cohort of Turin residents employed in waste treatment showed significant excesses for mortality, both for all causes (SMR = 162, 95% CI: 122–211) and for all cancers (SMR = 160, 95% CI: 102–239).

In conclusion the study confirmed the excesses for mortality and hospitalisation already detected in previous analyses, enlightening the responsibilities of the socioeconomic and occupational pattern, without excluding possible environmental effects, but reducing its size among the observed risks in previous studies.

A14.5 Conclusion

Current methods for spatial analysis of small areas studies are very useful in identifying sub areas in proximity of environmental sources of risks, like waste landfills. The studies carried out in Turin confirm risk excesses already detected in other studies. However, the causal relationship with the exposures due to waste landfill is unclear, firstly because of the ecologic type of the studies, secondly because many other risk factors, like industrial activities and socioeconomic factors, were not completely controlled by the study design and may have a huge role in the risk causation process.

A15. Incinerators, case study (Spain): Barcelona

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A15.1 Introduction

A total of 3 943 000 tons of municipal waste was generated in Catalonia in the year 2003. Of those, 892 540 tons (22.6%) were collected selectively, 745 211 tons (18.9%) were incinerated and 2 242 146 tons (56.9%) were dumped in controlled landfills. Around the year 2000, the individual waste generation in Catalonia was of 1.6 kg per person per day. It is generally reported that owing to improved emission control of waste incinerators and other industrial sources, levels of dioxins have steadily fallen in western Europe and the United States. Nevertheless, the attitude of public health authorities and the public has always been sceptical concerning potential health effects among residents who live in the vicinity of incinerators.

Emissions from MSWI include particles and gases (HCl, SO₂, nitric acid (HNO₃), etc.), metals (Cd, Pb, As, Hg, Cr etc.) and dioxins and other organochlorinated compounds. A large MSWI was installed in Mataró, a city located 25 kilometres north of Barcelona. The incinerator in Mataró has two furnaces and has been emitting (most of the time) within legal limits (0.1ng/m³). The waste management capacity is of 164 000 tons and since 1998; this capacity was insufficient to deal with all incoming waste. In 2005, about 21 000 tons of waste were sent to a landfill. The energy production was 84 443MWh/year and the energy self-consumption 11 090 MWh/year.

A15.2 Why did we do the study?

The study was requested by the municipality of Mataró following pressure from the community and fear for effects of dioxins. It should be noted that fear of dioxins was more prevalent in the communities in the mid 1990s.

The options we considered were four:

1. promoting emission control
2. evaluating the environmental impact
3. conducting a biomonitoring or biomarker based study
4. examining the adverse health effects (immediate or long term).

We opted for a biomarker based study since this responded best to the request we had to evaluate health-related effects and, in addition, it could be done in a reasonable time. A study on short- or long-term health effects was disregarded because of the minimal power it would have to detect any effect, particularly since our a priori hypothesis was that at current levels of emission if any effect did exist, this would be minimal and practically impossible to detect.

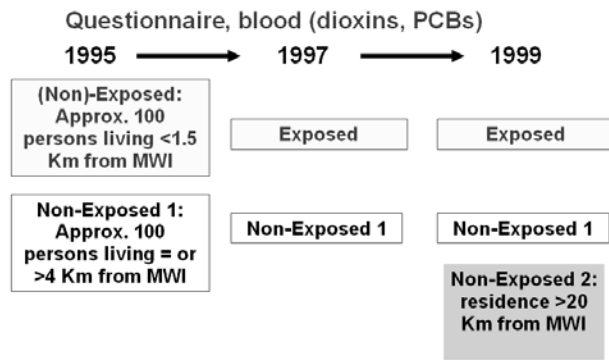
A15.3 Methods

We conducted a study comparing blood levels of dioxins when the plant started functioning in 1995, and two years after in 1997 (Figure A7). Participants in the study were 104 residents living within 1.5 km from the plant, and 97 residents, matched by age, sex and socioeconomic status, living at about 4 km from the plant. Subjects were between 18 and 69 years of age in 1995, of both sexes, and were randomly selected from the municipal list of inhabitants. A more-detailed description of this first phase of the study and results has been published (Gonzales et al., 2000, Gonzales et al., 2001). In 1999 we took a third blood sample and administered the same questionnaire on lifestyle factors and occupation, and repeated dioxin blood measurements in the same individuals of Mataró. We also included a new general population random sample from a similar city, but without any known source of dioxin exposure, located 20 km north of the incinerator.

In each phase we pooled the blood samples for the same comparative group of subjects and did a total of 38 analyses. The same WHO accredited laboratory (Olaf Päpke, ERGO Research laboratory, Hamburg, Germany) performed all analyses by gas chromatography-mass spectrometry. Standardized procedures were followed for blood collection, storage and shipping and duplicate measurements were done in stage II of stored samples from the

first phase of the study. Analyses were done for dioxins, furans and PCBs. Samples were pooled including about 20 subjects each and pooling was done on the basis of residence, age and sex. An additional sample was pooled for workers in the MSWI. Metals (Enrique Gadea, National Institute for Occupational Safety and Health (CNCT), Barcelona) were analyzed in blood/urine.

Figure A7. Study design, MSWI study Mataró



Source: Manolis Kogevinas, presentation made during the workshop

Municipal solid waste incinerators certainly contribute to overall environmental contamination with dioxins. Any health effects related to emissions from incinerators are not easily identified nor are they concentrated among residents close to the incinerator.

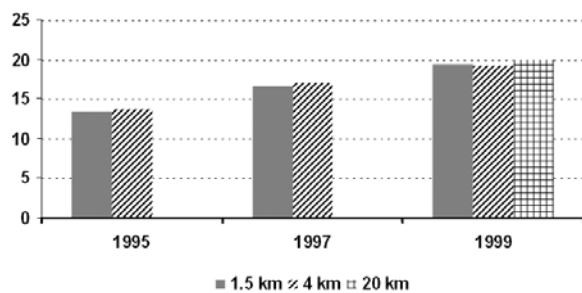
These effects may occur in the long term via general increase of environmental contamination. Decisions on waste management should take into account results of studies such the one in Mataró.

However, these decisions are essentially political decisions and should be based on wider considerations on waste management and long-term environmental contamination.

A questionnaire was completed by all participants requesting information on sociodemographic factors, lifestyle, occupation, medical history, nutrition (abridged food frequency questionnaire (FFQ)), consumption of locally grown foods, perception of smells from compost plant, reproductive history, nursing, weight, height measured, respiratory symptoms (second survey only, European Community Respiratory Health Survey (ECRHS) questionnaire).

A15.4 Results

Figure A8. Average dioxin levels (I-TEQ) in residents from Mataró (Barcelona), according to distance of their residence from the incinerator (1995, 1997, 1999)



We found a continuous increase of dioxin blood levels over the four years of the study (Figure A8). The increase was similar in people living near and far from the incinerator, affecting both sexes and all groups of age. Levels in Mataró in 1999 were similar to those of the group of people living in the comparison city. Both the results of the geographical comparison within Mataró and the comparison with the other city indicate that these increasing levels over time cannot be attributed to this particular incinerator. The current blood levels are around 20 pg I-TEQ/g fat. This level is low but still higher than the present levels of about 15 pg I-TEQ/g fat found in the general population of other developed countries.

Source: Manolis Kogevinas, presentation made during the workshop

A15.5 Discussion

The gradually increasing levels of dioxins in this population sample in Spain is surprising, since this trend in contrast to the general tendency in many industrialized countries. We considered potential explanations for this increasing trend including laboratory errors, aging of the population, exposure from incinerator (via local food consumption) and an increase due to general food exposure.

The laboratory doing the analyses is accredited and in addition duplicate analyses were conducted in the second phase that confirmed results. Aging of the population may have influenced the observed increase in Spain, although a 40% increase in 4 years cannot be explained solely by aging. Selection bias is not a tenable explanation because our sample consists of identical persons from the general population studied in consecutive time periods. In general population's diet is the most important source of exposure, contributing more than 90 % of the daily intake of dioxins. The observed increasing trend of dioxins blood levels in the Spanish population, and maybe in other countries, could reflect a recent increase of exposure to dioxins from foods or other unidentified sources.

The close contact of the study group with the municipality, ecological and other population groups and the links that existed already with these groups facilitated the presentation of the results and the acceptance of the main conclusion that the incinerator was probably not the main source for the increasing dioxin levels in the population.

A16. Incinerators, case study (France): dioxins emitted from a municipal solid waste incinerator and risk for non Hodgkin's lymphomas and soft tissue sarcomas. An ecoepidemiology case study in Besançon, France.

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A16.1 Background and study site

It is not clear whether low environmental doses of dioxin affect the general population. We have therefore conducted a multi-step research program around a MSWI with high emission levels of dioxin (Besançon, France).

The latter is located 4 km west of the city centre. Combustion chambers 1 and 2 (each with a capacity of 2.1 metric tons per hour) were put into service in 1971. In 1976, a third combustion chamber was opened (with a capacity of 3 metric tons per hour). In 1998, approximately 67 000 metric tons of waste were processed. Combustion chamber 1 (the most polluting) was shut down on December 31, 1998. Combustion chamber 2 was replaced by a new one with up-to-date pollutions controls (combustion chamber 4), which started operation in late 2003.

Some legal guidelines for incinerator emissions had not been followed at this location. For example, in 1997, dust and hydrogen emission levels were higher than prescribed and exhaust gases were not maintained at temperatures of more than 850 °C for the legal time (≥ 2 seconds), allowing dioxins to be emitted. The first time that the dioxin concentration of an exhaust gas was ever measured (in December 1997) it was found to be 16.3 ng international toxic equivalency factor (I-TEQ)/m³, whereas the European guide value is 0.1 ng I-TEQ/m³.

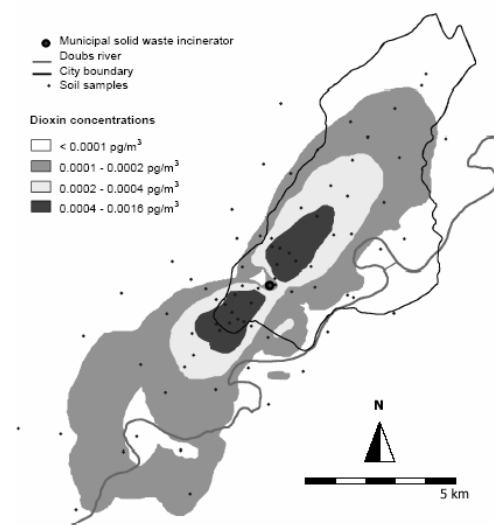
A16.2 Cluster detection

This ecoepidemiology approach started with a space–time clustering study using STS and NHL (which had been consistently associated with dioxin exposure) incidence data (Viel et al., 2000). Clusters were identified from 1980 to 1995 in the area ("departement") of Doubs by applying a space–time scan statistic to 26 electoral wards. The most likely and highly significant clusters found were identical for STS and NHL, made up of two electoral wards: Besançon (containing the plant, 114 000 inhabitants) and Audeux (west and contiguous to the former, 29 000 inhabitants). Standardized incidence ratios were 1.44 (observed number of cases = 45, focused test p value = 0.004) and 1.27 (observed number of cases = 286, focused test p value = 0.00003), respectively. We have also highlighted a space–time interaction for the more recent years in the same areas (an early time windows would have represented a counterargument for an association between the incinerator and the outcomes under study, since the time gap between the opening of the facility and such space–time clusters would have been too short considering the latency period). Confounding by socioeconomic status or urbanization seemed unlikely to explain the clusters. Moreover, the consistency of the results across genders (clusters involving only males would have favoured an occupational exposure), made us suspect an environmental pathway involving dioxin.

A16.3 Mixed individual/ecological case–control study

To further explore the environmental route suggested by these findings, we carried out a population–based case–control study at the Besançon city scale (excluding therefore 29 000 inhabitants of Audeux) since census data (needed to sample population controls, but only available at the block level – typically a quadrangle bounded by four streets) were available only for this area. We compared 222 incident cases of NHL diagnosed between 1980 and 1995 and controls randomly selected from the 1990 population census, using a 10-to-1 match (Floret et al., 2003). We used the same study design for the 37 incident cases of STS (Floret et al., 2004).

Dioxin ground-level air concentrations were modelled with a first generation Gaussian-type dispersion model (described in Figure A9), digitalized, contoured onto the surface of the map and classified in four categories of increasing exposure. Using Geographic Information System technology, case residences were linked to the dispersion map by exact address whereas control residence were at the block level.

Figure A9. The Gaussian-type dispersion model**A16.3.1 NHL results**

The risk of developing NHL was 2.3 times higher (OR = 2.3, 95% confidence interval (CI): 1.4–3.8) among individuals living in the area with the highest dioxin concentration than among those living in the area with the lowest dioxin concentration. No increased risk was found for the intermediate dioxin exposure categories (Table A7).

Table A7. NHL results

DIOXIN EXPOSURE	CASES	CONTROLS	OR (95% CI)
Very low	42	441	1.0
Low	91	952	1.0 (0.7–1.5)
Intermediate	58	681	0.9 (0.6–1.4)
High	31	486	2.3 (1.4–3.8)

Adjustment for a wide range of socioeconomic characteristics at the block group level did not alter the results.

Although emissions from incinerators are usually not regarded as an important source of exposure to dioxins compared with other background sources (intake from food is considered to account for over 90% of the burden of dioxins in the general population), these findings support the hypothesis that environmental dioxins increase the risk of NHL among the population living in the vicinity of a MSWI.

A16.3.2 STS results

Compared with the least exposed zone, the risk of developing STS was not significantly increased for people living in any of the more exposed zones (Table A8). We therefore concluded that the over incidence observed in the city of Besançon was not spatially organized according to the transport and dispersion of dioxin emissions.

Table A8. STS results

DIOXIN EXPOSURE	CASES	CONTROLS	OR (95% CI)
Very low	5	61	1.0
Low	15	156	1.2 (0.4–3.4)
Intermediate	15	126	1.4 (0.5–4.1)
High	2	27	0.9 (0.2–5.1)

However, the small size of populations living in the different dioxin exposure areas and the low STS incidence, limited considerably the statistical power and the likelihood of detecting any moderate increase of the risk.

Before concluding to the absence of STS risk, a nationwide multi-incinerator study, including a large number of subjects and increasing statistical power, would be necessary.

A16.4 Dioxin exposure assessment

The main limitation of the previous studies laid within the use of a first-generation Gaussian-type dispersion model as a proxy for dioxin exposure, since its accuracy had not been assessed before. We have therefore validated this geographic-based exposure through PCDD/F measurements from soil samples (Floret et al., 2006). Seventy-five sampling points were determined in relation to homogeneous geological and topographical conditions. PCDD/F concentration, pH, organic carbon concentration, cation exchange capacity, and geomorphology and ecology features were assessed for each soil sample.

PCDD/F soil concentrations ranged from 0.25 to 28.06 pg WHO-TEQ/g dry matter. An interaction between measured dioxin concentrations and topography complexity (simple terrain on the northeast side with gentle hills of moderate slope and complex terrain on the southwest side with more pronounced hills and valleys) was found (Table A9).

Table A9. Means (standard deviations) of dioxin soil concentrations (WHO-TEQ/g dry matter) per geographic-based exposure and topography complexity categories

GEOGRAPHIC-BASED EXPOSURE	VERY LOW	LOW	INTERMEDIATE	HIGH
Complex topography	1.09 (1.76)	2.44 (3.53)	1.91 (1.12)	1.37 (0.21)
Simple topography	1.81 (1.14)	1.99 (1.37)	3.53 (2.30)	11.25 (12.39)

These results were confirmed by multivariate models, adjusting for organic carbon concentration and altitude. In simple terrain, a significant association was highlighted between modelled dioxin ground-level air concentrations and log-transformed measured dioxin soil concentrations with a strong gradient across exposure categories. Conversely, in a complex topography situation, the model overpredicted ground-level air concentrations, particularly in the high exposure zone. First-generation modelling provided a reliable proxy for dioxin exposure in simple terrain, reinforcing the results of our NHL case–control study (since 89.5% of cases and 90.7% of controls lived on the northeast side). A logistic regression restricted to cases and controls residing on the northeast side yielded a slightly increased OR in the highest dioxin exposure area (OR = 2.5, 95% CI: 1.4–4.5), compared to our initial finding (OR = 2.3, 95% CI: 1.4–3.8). However, a more advanced atmospheric diffusion model should have been used for refined assessment in complex terrain. This study confirms that use of first-generation models should not be transferred to another geographic region without validation with measurement data from the new area.

A16.5 Individual case–control study

The mixed individual/ecological case–control study suffered two limitations. First, it was limited by the scarcity of covariates (only age and gender), which could potentially confound the relationship between dioxin exposure. Second, to circumvent the lack of actual exposure data, we used dispersion modelling as a proxy for dioxin exposure, assuming that residents within a given contour were homogeneously exposed and ignoring any occupational, residential or nutritional history (which could have allowed a more relevant exposure assessment).

We therefore launched a case–control study in which dioxins are measured in biological tissues. All incident cases, that accrued in this area during the 2003–2006 time period, are from now on included (40 cases), while a few controls are still to be recruited. Results are due within one year.

A17. Incinerators, case study (Portugal): environmental Health Surveillance related to waste incineration

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A17.1 Background

In response to public and scientific concerns that led to the closure of several solid waste incinerators (SWI) in Portugal during the past decade, the responsible entities may be obliged, following environmental impact studies, to conduct environmental and public health monitoring programs as risk surveillance and control measures. These programs generally address quality of air, soil, water and terrestrial ecosystems, as well as monitoring of human health status, health conditions and psycho-social factors associated with potential negative impacts of the incinerator.

The Institute of Preventive Medicine of Lisbon's Faculty of Medicine, through its Environmental Health Unit, was invited to monitor public health in relation to two incinerators: one in the metropolitan area of North Lisbon and another on the island of Madeira. To fulfil these requests, a model for surveillance was designed and an environmental health surveillance program (ProVEpA) relative to each facility was launched, with the main goals of monitoring the magnitude as well as spatial and temporal trends of specific indicators of both exposure to the environmental agents of greater concern and associated adverse health effects.

A17.2 Methodological strategy

The most relevant pollutants potentially emitted during the incineration processes include some of the so-called "classical" pollutants (for example particulate matter as well as acid and oxidant pollution) and the micropollutants, comprising dioxins and heavy metals and considered the most critical in terms of public concern. Pathologies and health conditions such as asthma, cancer, reproductive and mental health disorders, neurobehavioral dysfunctions, and alterations in global health status and well-being raise particular scientific and public concerns as they have been shown or are suspected to be associated with human exposure to those pollutants.

HBM is gaining ever greater recognition as an essential tool in assessing population exposure to chemical environmental agents (Angerer, 2002). Much more so than by analysis of environmental quality, biomonitoring in humans provides a possibility to directly assess human exposure to environmental pollutants and the potential health effects of such exposure. It is a reliable source of information when linked to toxicological and environmental monitoring data. Moreover, biomonitoring takes into consideration the differences between individuals with regard to uptake, which are due to differences in hygiene, breathing characteristics, etc., as well as metabolism and elimination. On the other hand, by measuring the internal dose of chemicals biomonitoring accounts for exposure from all sources, pathways and routes of absorption. Therefore, it has several advantages over environmental monitoring for large-scale survey programs when the objective is to monitor human exposure to relevant chemicals and to determine spatial and temporal trends of exposure to those chemicals (Casteleyn, 2004).

Acknowledging the relevance of HBM, European Commission is endorsing it (ESBIO, 2005) under Action Three of its Environment & Health Action Plan 2004–2010 (European Commission, 2004).

Within this increasing importance of biomonitoring as an environmental health surveillance tool, each ProVEpA includes HBM projects focused on heavy metals and on dioxins and dioxin-like compounds, besides several epidemiological investigations on the pathologies and health conditions associated with human exposure to the most relevant pollutants. It also includes monitoring of self-perception of the health risk due to residing near the incinerator.

Two different population groups in terms of distance of residence from the incineration facility are included in each monitoring Project, ensuring that they are as much as possible similar in all other characteristics. Moreover, data is collected longitudinally, in order to provide information on temporal trends of the relevant exposures and of associated health effects in relation to a baseline reference situation, before regular operations started. Because other environmental exposures not related to incinerator operation (namely exposure to environmental tobacco smoke (ETS)) can adversely affect health in the same way as the pollutants emitted during incineration processes, monitoring of ETS as a potential confounding factor is also considered.

Three different and complementary components are therefore included in the adopted surveillance model:

- biological survey: HBM addressing exposure to the most critical pollutants, namely heavy metals and dioxins;
- adverse health effects survey: monitoring of the most relevant adverse health effects associated with incineration processes; and
- risk factors survey: monitoring of tobacco consumption.

Depending on selected study populations and addressed biomarkers or pathologies and health conditions, several monitoring projects can be identified: within the biological survey, dioxins are monitored in the blood of the general population and in human milk; heavy metals are measured in the blood of the general population; lead and mercury in maternal and umbilical cord blood, as well as in mothers' pubic hair, of selected mother–newborn pairs; and in blood and scalp hair of children between one and six years old. The adverse health effects survey includes monitoring of the prevalence among young students of asthma and among general population adults of: cancer incidence and mortality, global and mental health status and frequency of reproductive and developmental disorders. The risk factors survey further addresses prevalence of tobacco consumption in the same adolescents as the asthma monitoring project.

A17.3 Results

So far, at least three (and up to seven) observations have already been performed in each monitoring project included in both ongoing ProVEpAs related to waste incinerators near Lisbon and in Madeira.

Results obtained so far indicate that exposure to dioxins (Reis et al., 2007a, Reis et al., 2004a, Reis et al., 2007b, Reis et al., 2004b) and heavy metals (Reis et al., 2007c, Reis et al., 2007d, Reis et al., 2007e) is adequately controlled, as assessed by HBM of these pollutants in tissue and body fluids of the different study groups in the two areas. In general no statistical differences were found between exposed and control groups and strong similarities were observed in the distributions and means of anthropometric, sociodemographic, lifestyle, behavioural and, for women, maternal–related variables characterizing the groups.

Comparing corresponding results from Lisbon and Madeira, namely dioxins in blood of general population adults and in human milk, levels in Lisbon are, in general, higher than in Madeira, indicating a higher general exposure profile which appears to be independent of incinerator influence.

In terms of the monitoring of adverse health effects and risk factors, asthma prevalence in adolescents has neither shown any differences between exposed and control areas nor varied over the years of monitoring. No differences in smoking patterns were observable between exposed and control areas when adjusted for age and gender. Attention must be drawn to the extraordinary potential of the results in terms of public health in that they characterize the breadth of impact of this factor known to constitute a serious risk to the health of adolescents, thus offering an opportunity for prevention that should not be wasted.

Concerning the other surveys in the program, monitoring of additional adverse health effects showed that cancer mortality and morbidity in the study areas and in the baseline and first study periods considered remained practically stable. The apparent temporal and spatial variability in some rates is not clinically relevant and could be explained by the reduced number of cases. However, the evidence that, in the case of cancer, the effect may occur many years after the exposure points to the need to continue to investigate potential associations between SWI activity and indicator trends.

Trends in infant mortality due to all causes of death have shown a general decrease, reflecting a global tendency. Monitoring of other reproductive alterations between the baseline and the first years of surveillance does not yet allow any conclusions to be drawn.

A longitudinal study among Madeira's population, involving interviews to study self–perception of state of health and of quality of life related to health factors as well as mental well-being, as assessed by questionnaire and including evaluation of anxiety and depression prevalence, found no significant differences between control and exposed groups.

A17.4 Conclusions

In general terms, some conclusions can already be drawn from the results obtained.

Differences between exposed and control populations were overall not statistically significant for exposures, pathologies or health conditions, suggesting the effectiveness of source control measures in operation in both incinerators under study. Furthermore, results from exposure determinations show levels generally below known

“reference” values and lower than those determined during the baseline situation, presenting evidence for a general significant trend for reduction of exposure to all studied pollutants. Levels in Lisbon are, on a whole, higher than those for Madeira, indicating a lower exposure to critical pollutants and less related health risks in Madeira, which is more likely to be attributable to greater industrialization as a whole in Lisbon.

To conclude, the importance of developing ProVEpAs is apparent both from the perspective of verifying the adequacy of incinerator process controls and from the issues raised that require further study over a longer time-period or more in-depth research. The projects allow serious gaps in relevant information to be filled in, even if only at local level, thus identifying relevant aspects susceptible of prevention and providing information on which policy decisions can be based.

Acknowledgements

The programs referred to are developed through the efforts of a large scientific and technical team, each of whom the authors are greatly indebted to.

A18. Waste treatment and health in Campania, southern Italy

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A18.1 Introduction

For more than a decade, waste management in the region of Campania, in southern Italy, has been problematic. The area has been described as the “Triangle of death” (Bianchi et al., 2004, Senior & Mazza, 2004), and repeated episodes of social unrest take place on an almost regular basis. Indeed, waste in the region is poorly managed and existing facilities are insufficient: landfills are exhausted, recycling is insufficient, quality of refuse derived fuel produced by seven plants is poor, one incinerator is being built.

The health impact of waste is largely unknown, although claims are often made that large excess in mortality and morbidity observed in the region are attributable to waste exposure (Altavista et al., 2004, Minichilli et al., 2004).

World Health Organization was charged by national authorities examine the health implications of widespread uncontrolled waste dumping and toxic waste burning. A collaborative project carried out with ISS, National Research Council and health and environmental regional authorities was set up in 2004 to investigate mortality and morbidity in the area most affected (the provinces of Naples and Caserta) and its association with exposures connected to waste.

A18.2 Methods

A first descriptive phase of the epidemiological study has been completed and results have been published (Comba et al., 2006). The two provinces were described with regards to cancer mortality and congenital malformation profile, for the period of time 1994–2002. The population was around 4 million people. Mortality records were retrieved through national statistics and congenital malformation data through Regional Birth Defects Registry. Twenty causes of deaths (all cause mortality, all cancer mortality and specific cancer causes) and 11 types of congenital malformation were studied across 196 municipalities. Traditional and Bayesian methods were used to describe the geographical distribution of risks. SMRs (95% CIs) and BMRs (95% uncertainty intervals), calculated as suggested by Besag, York and Mollié (1991), were computed with regional reference; the risk estimates were then mapped to describe the geographical distribution of risks at municipality level and to investigate on the portion of variability due to spatial structure (Mollié, 2000).

Given the lack of information on specific exposures linked to waste, a mixed criterion, defined post hoc, was applied for identifying the municipalities most at risk, as follows. Municipalities were ranked based on the number of statistical significant excesses; significance was reached in when:

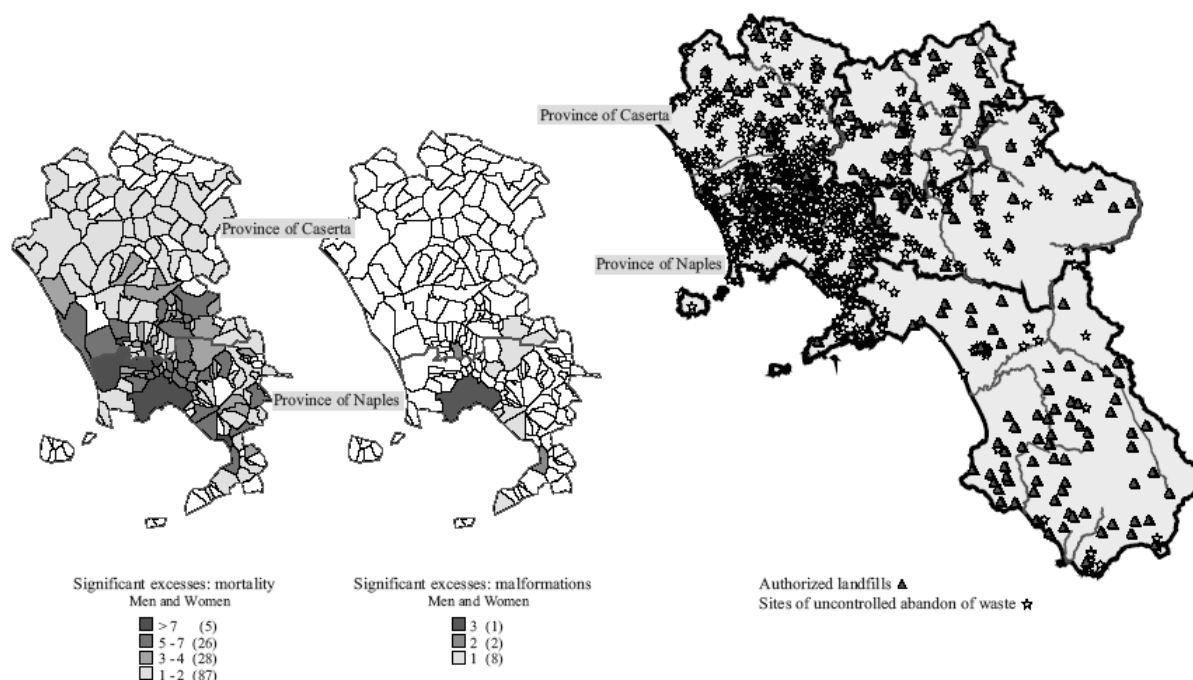
- both the SMR and the BMR were statistically significant; or
- the BMR was statistically significant and the observed number of cases was equal or greater than three.
- In parallel, a dialogue was initiated on whether and how a health impact assessment of the available policy options can be launched.

A18.3 Results

The two provinces are more at risk than the reference region for a wide set of cancer causes. A group of about 40 municipalities have consistent, repeated and significant excess for several of the health outcomes considered, including cancer of the stomach, kidney, liver and lung and for urogenital and cardiovascular congenital malformations.

The majority of the excess was observed in an area between the two provinces, where most of the illegal waste treatment activities are known to take place, as visible in Figure A10.

Figure A10. Municipalities with risk excesses for mortality (first map, 1994–2001) and congenital malformations (second map, 1996–2002) and geographical dislocation of legal and illegal waste disposal sites (third map)



A18.4 Conclusion

Using established methods for spatial analysis of small areas studies allowed the identification of a sub area in which large and significant excess were observed for several health outcomes. This confirms that concerns about the health impact of environmental factors are founded. However, the link with the exposures due to waste disposal is unclear, firstly because of the lack of good exposure information, secondly because many other risk factors, widespread industrial activities and a very high population density, are at play.

A second phase of the study, aiming at clarifying the possible causal nature of the association, is underway. It will include:

1. the creation of a “waste risk” indicator at municipality level based on an accurate georeferenced examination of the water, air and soil contamination to study the correlation of adverse health effects with environmental exposures;
2. the inclusion of confounding socioeconomic factors in the analyses;
3. the individuation of the clusters of municipalities more at risk; and
4. the establishment of a risk communication group, in response to public concern about the additional environmental load that could be brought in the area by new waste disposal facilities whose health impact would need to be assessed and monitored.

Results will be published in the next few months.

With regard to HIA, a network of representatives of local health authorities and citizens’ interest groups was established. HIA was proposed, and accepted in principle, as a means to incorporate health considerations in the process. However limited flexibility in the concrete options for waste management creates some difficulties. Clearer identification of specific objectives and methods of work seems to be needed before embarking in a full-fledged HIA.

Annex B. Programme

Thursday, 29 March 2007

09:00	<i>Registration</i>
09:30–9:40	Welcome and introduction; nomination of chairperson (WHO)
09:40–10:10	Overview on waste and health effects: landfills (P. Saunders and H. Dolk)
10:10–10:40	Overview on waste and health effects: incinerators (A. Staines)
10:40–11:00	Technology of waste management and exposure assessment: landfills (L. Musmeci)
11:00–11:30	<i>Coffee break</i>
11:30–11:40	Case studies: introduction (P. Comba)
11:40–12:00	Case study 1: Denmark (H. Hansen)
12:00–12:20	Technology of waste management and exposure assessment: incinerators (G. Viviano)
12:20–12:40	Case study 2: Barcelona (M. Kogevinas)
12:40–13:00	Discussion
13:00–14:00	<i>Lunch break</i>
14:00–14:40	Case study 3: UK (D. Briggs)
14:40–15:20	The public health role in waste management (P. Davies)
15:20–15:40	Case study 4: Turin, Italy (E. Cadum)
15:40–16:00	<i>Coffee break</i>
16:00–16:30	HIA of waste management facilities (I. Matthews)
16:30–16:50	Case study 6: Besançon, France (J. F. Viel)
16:50–17:10	Priority needs in research (F. Bianchi)
17:10–17:30	Discussion

Friday, 30 March 2007

09:30–09:50	Public views on sources of knowledge for decisions about waste management (A. Staines)
9:50–10:20	Monetary valuation of impacts and cost–benefit analysis (A. Rabi)
10:20–10:40	Case study 5: Portugal (F. Reis)
10:40–11:00	The industry's view (R. Caggiano)
11:00–11:30	<i>Coffee break</i>
11:30–11:50	The NGO's view (S. Ciafani)
11:50–12:10	Risk assessment of waste management policies: the INTARESE project (F. Forastiere)
12:10–12:30	Supporting decision-making in waste management: the Campania Study (M. Martuzzi)
12:30–13:00	Conclusions and recommendations
13:00	<i>Closure</i>

Annex C. List of participants

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