Review

A review of waste management practices and their impact on human health

L. Giusti *

Faculty of Health and Life Sciences, UWE Bristol, Frenchay Campus, Coldharbour Lane, Bristol BS16 1QY, United Kingdom

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ABSTRACT

This work reviews (i) the most recent information on waste arisings and waste disposal options in the world, in the European Union (EU), in Organisation for Economic Co-operation and Development (OEDC) countries, and in some developing countries (notably China) and (ii) the potential direct and indirect impact of waste management activities on health. Though the main focus is primarily on municipal solid waste (MSW), exposure to bioaerosols from composting facilities and to pathogens from sewage treatment plants are considered. The reported effects of radioactive waste are also briefly reviewed. Hundreds of epidemiological studies reported on the incidence of a wide range of possible illnesses on employees of waste facilities and on the resident population. The main conclusion of the overall assessment of the literature is that the evidence of adverse health outcomes for the general population living near landfill sites, incinerators, composting facilities and nuclear installations is usually insufficient and inconclusive. There is convincing evidence of a high risk of gastrointestinal problems associated with pathogens originating at sewage treatment plants. In order to improve the quality and usefulness of epidemiological studies applied to populations residing in areas where waste management facilities are located or planned, preference should be given to prospective cohort studies of sufficient statistical power, with access to direct human exposure measurements, and supported by data on health effect biomarkers and susceptibility biomarkers.

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* Tel.: +44 (0)117 3283543; fax: +44 (0)117 3282904.
  E-mail addresses: Lorenzo.Giusti@uwe.ac.uk, lacinaevicina@yahoo.co.uk.

1. Introduction

Human activities have always generated waste. This was not a major issue when the human population was relatively small and nomadic, but became a serious problem with urbanisation and the growth of large conurbations. Poor management of waste led to contamination of water, soil and atmosphere and to a major impact on public health. In medieval times, epidemics associated with water contaminated with pathogens decimated the population of Europe and even more recently (19th century), cholera was a common occurrence. Some of the direct health impacts of the mismanagement of waste are well known and can be observed especially in developing countries.

As science and technology developed, the management of an ever increasing volume of waste became a very organised, specialised and complex activity. The characteristics of waste material evolved in line with changes in lifestyle, and the number of new chemical substances present in the various waste streams increased dramatically. The long-term health effects of exposure to substances present in the waste, or produced at waste disposal facilities are more difficult to measure, especially when their concentrations are very small and when there are other exposure pathways (e.g. food, soil). Nonetheless, lack of evidence can cause public concern. Well-publicised industrial accidents, often unrelated to waste management activities, have produced a NIMBY (not in my backyard) syndrome that causes fierce opposition to the construction of landfills, incinerators, or other waste disposal facilities. Government and health authorities are under increasing pressure from the public to provide epidemiological evidence of potential adverse health effects produced by these activities. Thousands of manuscripts have been published on the impact of emissions in proximity of waste disposal sites. A number of authors have written reviews, and reviews of reviews. Epidemiological studies have often shown the existence of an association between human illnesses and proximity to a waste disposal site, or length of residence near such site, but the overwhelming majority have failed to provide significant evidence of a causal link.

The main aims of this review are the following:

(i) summarise the most recent information on waste arisings and waste disposal options in the world, in the European Union (EU), in Organisation for Economic Co-operation and Development (OEDC) countries, and in some developing countries;
(ii) evaluate the epidemiological evidence of direct and indirect impact of waste management activities on health.

The main focus is on municipal solid waste (MSW), but composting facilities and sewage treatment plants are also considered. Also, the results of epidemiological studies on the effects of exposure to radioactive waste are briefly reviewed.

This study builds on the work carried out by Saffron et al. (2003). The literature search was carried out using the same online databases and included primary studies and reviews of epidemiological investigations. The quality of the studies was classified on the basis of the following criteria:

(i) epidemiological study design (experimental studies and prospective cohort studies were listed at the top of the hierarchy);
(ii) sample size and statistical power of the study;
(iii) consideration of confounding factors (such as other sources of pollutants both indoors and outdoors);
(iv) availability of exposure data (as opposed to using surrogates such as distance from waste management facilities, or post code of residence);

(v) inclusion of information on waste management procedures at each site (as this can affect the level of a pollutant, its pathways, and exposure route);
(vi) studies carried out on human population (as opposed to studies on animals);
(vii) the strength of the relationship found between possible cause and effect, based on the reported 'relative risk'.

General information on the different types of epidemiological studies is also provided for readers unfamiliar with the research methodologies used in this field.

2. Waste production

2.1. Municipal solid waste (MSW)

The mass of waste produced in the world has been growing considerably for many decades especially in affluent countries as shown by the link between national gross domestic product (GDP) and waste generation per capita (World Bank, 1992; OECD, 2003). Though waste data on waste arisings is often incomplete and in some cases unreliable, recent estimates suggest that the municipal solid waste (MSW) alone generated globally exceeded 2 billion tonnes per year at the turn of the millennium (e.g. Key Note, 2007).

In 2006, the USA produced more than 228 million tonnes (EPA, 2008; OECD, 2008a,b) of MSW, or 750 kg per capita. The quantity of MSW generated in the OECD area in 2006 was more than 619 million tonnes, or 580 kg per inhabitant (OECD, 2008b). Fig. 1a and b shows the MSW arisings in selected OECD countries and in China, as total weight and as kg/yr/capita, respectively.

In 2006, the 15 countries of the European Union (Austria, Belgium, Denmark, Finland, France, Germany, Greece, Italy, Ireland, Luxembourg, Netherlands, Portugal, Spain, Sweden, UK) generated 219 million tonnes of MSW, or 560 kg/yr/capita (OECD, 2008a,b).

As less developed countries such as China and India industrialise and their populations urbanise, huge amounts of municipal waste are disposed of, though the production per capita (less than 0.5 kg/day/capita in India and less than 0.9 kg/day/capita in China) is still relatively small compared to the production in most individual OECD counties (up to 2.1 kg/day/capita in the USA). However, this masks the fact that a large proportion of the MSW is produced
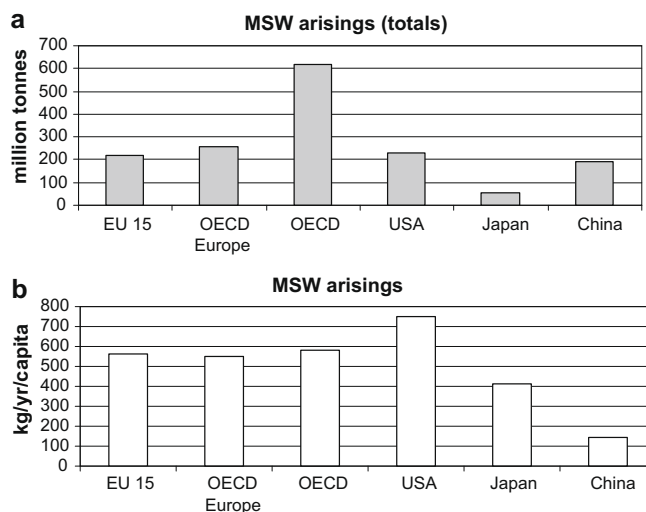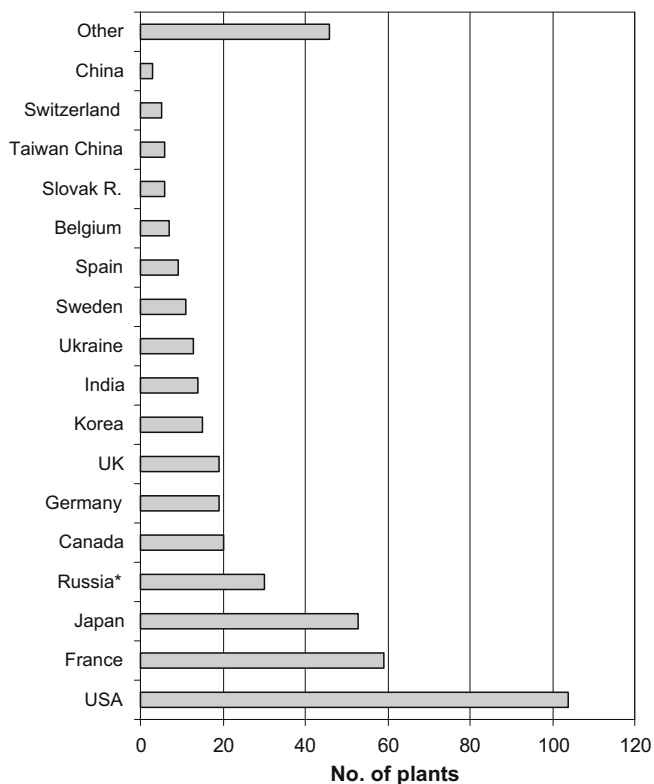

Fig. 1. (a) MSW arisings in selected OECD countries and in China, in million tonnes per year, and (b) MSW arisings in selected OECD countries and in China, as kg/yr/capita. Sources of data: OECD, 2008a; OECD, 2008b; EASUR, 2005.

## Operating nuclear power plants in the world in 2008

Fig. 2. Distribution of nuclear power plants operating in the world in 2008. Source of data: McDonald, 2008.

* High uncertainty about these data

Annual spent fuel arisings from nuclear power plants

* High uncertainty about these data

Fig. 3. Annual spent fuel arisings (in tonnes of heavy metals) in countries that have a large number of nuclear reactors. Sources of data: OECD, 2008b; IAEA, 2007; Li, 2001.

in urban centres. In 2002, more than 1 billion tonnes of industrial waste (about five times the amount of MSW) was produced in China, mostly mine tailings, coal ash, and slag, and by 2030 China is expected to generate approximately twice as much municipal waste as the USA, while India will overtake the USA (EASUR, 2005). The published projections of municipal waste generation for China were based on three different waste growth scenarios (i.e. waste generation increasing gradually from 0.9 kg/day/capita to 1.2 kg/day/capita, 1.5 kg/day/capita, and 1.8 kg/day/capita). Even assuming a low waste generation scenario, the total amount of MSW generated in 2030 would be close to twice the waste predicted to be produced in the USA. Though the GDP growth rate for China is no longer in double digits, the global economic downturn is unlikely to affect the projected relative waste production of these countries.

2.2. Radioactive waste

Radioactive waste and ionising radiation are produced at every step of the nuclear fuel cycle, starting from mining and mineral processing, through uranium enrichment, fuel rod fabrication and reprocessing, to nuclear power generation, and the decommissioning of nuclear power plants. Weapon production in the military sector is another major source of radioactive waste. According to 2007 estimates (IAEA, 2007), there are about 5.5 million tonnes identified uranium resources, and the total world production is about 40,000 tonnes of uranium (39,600 tonnes in 2006), with major producers including Canada (25%), Australia (19%), Kazakhstan (13%), Niger (9%), and the Russian Federation (8%). The annual demand of uranium by the nuclear industry is around 67,000 tonnes, so the additional demand of uranium is balanced by supplies from
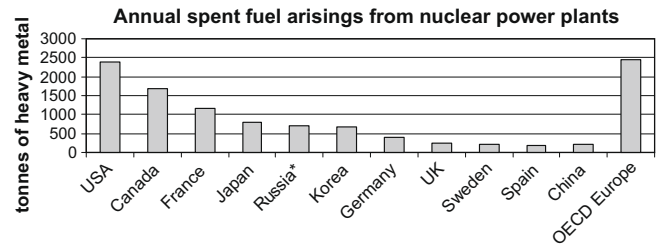
other sources, particularly military sources and spent fuel reprocessing plants.

Nuclear energy presently provides about 15% of the world's electricity, almost 24% of electricity in OECD countries, and 34% in the EU (EIA, 2008). In Europe, France is the largest producer (about 42% of all EU electricity), followed by Germany (14%) and the UK (9%). Thirty countries are members of the OECD, including 20 of the 27 EU states, Canada, the USA, Japan, Australia, New Zealand, Mexico, Switzerland, Turkey, Iceland, and South Korea. Fig. 2 summarises the distribution of nuclear power plants in the world, and Fig. 3 shows the annual spent fuel arisings in countries that have a large number of nuclear reactors and in a few others listed for comparative purposes. Of the 439 plants operating in 2008, 104 were in the USA, 59 in France, 55 in Japan and 31 in Russia. The main planned expansion of nuclear power production is based in Asia, where 20 plants are presently under construction, of which 6 in China and 6 in India (McDonald, 2008). In addition to the operating power reactors, at least 110 reactors shut down (IAEA, 2006) and the majority of the operating plants have been in production for more than half their planned life. Up to 2007, only 10 power reactors in the world had been completely decommissioned and their sites released for unconditional use, others have been partially dismantled, but the majority still require decommissioning (IAEA, 2007). Other types of nuclear facilities will also require decommissioning, such as reprocessing plants, uranium enrichment plants, and nuclear submarines.

About 10,000–12,000 tonnes of heavy metals of spent fuel material are produced annually. So far, just over 30% of the 290,000 tonnes of heavy metals discharged from commercial nuclear power plants has been reprocessed to recover uranium and plutonium (WNA, 2008). The world commercial reprocessing capacity is 5550 tonnes of heavy metals per year and most of the spent fuel is kept in storage facilities. The main reprocessing facilities are at La Hague (France), Sellafield (UK) and Mayak (Russia).

3. Waste management practices

A number of serious and highly publicised pollution incidents associated with incorrect waste management practices, led to public concern about lack of controls, inadequate legislation, environmental and human health impact. This in turn forced many national and federal governments to introduce new regulatory frameworks to deal with hazardous and unsustainable waste management operations. A waste management hierarchy based on the most environmentally sound criteria favours waste prevention/minimisation, waste re-use, recycling, and composting. In many countries, a large percentage of waste cannot presently be re-used, re-cycled or composted and the main disposal methods are landfilling and incineration.

In Europe, landfilling is the main disposal method. In 1999, 57% of MSW was landfilled (67% in 1995) in western Europe, and 83.7%
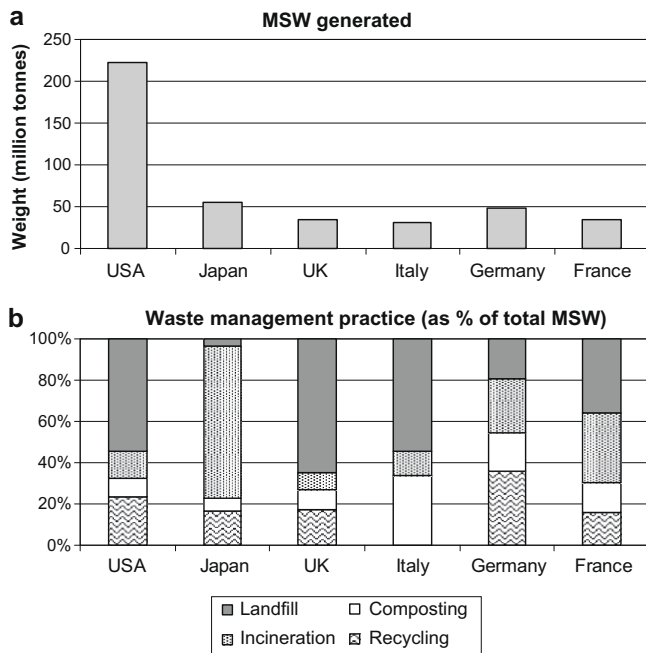
Fig. 4. (a) MSW generated in selected countries, and (b) contrasting waste management practice in selected countries. Source of data: OECD, 2008b.

in central and eastern Europe (DHV CR, 2001). In 2000, about 18% of MSW was incinerated and 25% recycled in western Europe, whereas incineration and recycling accounted for 6% and 9%, respectively, in central and eastern Europe (Eurostat, 2002). Overall, recycling is increasing in western Europe. Lack of data makes it difficult to identify trends for eastern Europe.

In 2006 the USA landfilled 54% of MSW, incinerated 14%, and recovered, recycled or composted the remaining 32% (EPA, 2008). Fig. 4a and b shows the weight of MSW generated and some contrasting examples of waste management practice in different countries. The data for Japan refers to 2003, for Germany to 2004, and for UK, France, Italy and the USA to 2005. The percentage of MSW disposed at landfills accounted for 3% in Japan, 18% in Germany, 36% in France, 54% in Italy and the USA, and 64% in the UK. As legislation becomes more stringent, and landfilling becomes a less cheap option, alternative solutions are considered. For example, there has been a significant reduction in the amount of waste landfilled in the UK and Italy. In 1995, Italy landfilled 93% of MSW, and the UK 83%.

It is worth mentioning that a wide range of waste materials (sewage sludge, industrial waste) is increasingly spread on agricultural land as soil amendments. These undoubtedly produce a number of positive effects on soil quality, but also raise concern about potential short-term (e.g. pathogen survival) and long-term effects (e.g. accumulation of heavy metals). Climate change will also become a major incentive to the use of biosolids on agricultural land, especially in regions where longer periods of low rainfall and mean higher temperatures are expected. In many parts of the world (e.g. Europe, USA) agricultural soils receive large volumes of soil amendments. Approximately 5.5 million dry tonnes of sewage sludge are used or disposed of annually in the United States and approximately 60% of it is used for land application (NRC, 2000a). The application of biosolids to soil is likely to increase as a result of the diversion of waste away from landfill sites, and due to increasing cost of artificial fertilisers (UNEP, 2002; Epstein, 2003).

The type of waste management practices adopted in each country are mostly a function of economic considerations, but are also a reflection of technical aspects due to the type of waste to be han-

dled. For example, if houses and buildings are heated by coal burning, large amounts of coal ash may end up disposed together with other urban waste. As coal ash contains high concentrations of heavy metals and other potential contaminants, this type of mixed urban waste cannot be easily composted. Coal ash also makes incineration less efficient. A change of energy source from coal to gas can thus have important beneficial effects on waste management options. This is important for many developing countries. Landfilled putrescible waste causes gas and leachate production. In Europe, the EU Directive 1999/31/EC on the landfill of waste has stimulated the diversion of organic matter to composting or specialised landfill sites, especially in the Netherlands, Sweden, Denmark, and Austria. Incineration is not an option for organic material due to its water content. Waste separation at source allows the removal of hazardous (flammable, toxic) items, better recycling and composting options, and a reduction of MSW to be disposed of. Therefore, knowledge of waste composition is of vital importance for the choice of waste treatment and disposal.

4. Health issues

Despite important technological advancements, improved legislation and regulatory systems in the field of waste management, and more sophisticated health surveillance, the public acceptance of the location of new waste disposal and treatments facilities is still very low due to concern about adverse effects on the environment and human health. Health issues are associated with every step of the handling, treatment and disposal of waste, both directly (via recovery and recycling activities or other occupations in the waste management industry, by exposure to hazardous substances in the waste or to emissions from incinerators and landfill sites, vermin, odours and noise) or indirectly (e.g. via ingestion of contaminated water, soil and food).

In the past, the performance of a large number of landfills and incinerators has been quite poor, including landfills that were built with a containment barrier (a clay liners or a synthetic membrane). Roche (1996) pointed out that the frequency of landfill failure in the UK was quite high, resulting in surface and groundwater pollution, despite the fact that about one third of 4000 sites surveyed had a clay liner. As a result of these technical failures, the public has developed a mistrust of the opinions of politicians and technical advisors. Plans for the construction of a new waste disposal facility or treatment plant normally meet fierce opposition from the local community due to the fear of potential adverse health effects, the association of these facilities with odours, noise, visual intrusion, and the reduction in value of land and property. Table 1 shows a simplified summary of the main known emissions and environmental impacts of waste management activities associated with MSW. A full assessment of environmental impact should also consider the positive effects of these activities to the local and wider community, especially in the case of modern, properly managed facilities. A similar table could be drawn for the direct and indirect health impacts of each waste management activity, but the range of possibilities is so large that any summary would be incomplete and subjective. The main pathways of exposure are inhalation (especially due to emissions from incinerators and landfills), consumption of water (in the case of water supplies contaminated with landfill leachate), the foodchain (especially consumption of food contaminated with bacteria and viruses from landspreading of sewage and manure, and food enriched with persistent organic chemicals that may be released from incinerators). It is also important to remember that occupational accidents in the waste management industry can be relatively common, higher than national average for other occupations (HSE, 2004), and often higher than the potential cases of adverse effects to the resident population investigated by epidemiological studies.

Table 1
Main environmental impact of municipal solid waste management.

	Water	Air	Soil	Landscape	Climate
Activity					
Landfilling	Leachate (heavy metals, synthetic organic compounds)	CO_2, CH_4, odour, noise, VOCs	Heavy metals, synthetic organic compounds	Visual effect, vermin	Worst option for greenhouse gases emission[a]
Incineration	Fall-out of atmospheric pollutants	SO_2, NO_x, N_2O, HCl, HF, CO, CO_2, dioxins, furans, PAHs, VOCs, odour, noise	Fly ash, slags	Visual effect	Greenhouse gases[a]
Composting	Leachate	CO_2, CH_4, VOCs, dust, odour, bioaerosols	Minor impact	Some visual effect	Small emissions of greenhouse gases[a]
Landspreading	Bacteria, viruses, heavy metals	Bioaerosols, dust, odour	Bacteria, viruses, heavy metals, PAHs, PCBs	Vermin, insects	Small emissions of greenhouse gases.
Recycling	Wastewater	Dust, noise	Landfilling of residues		Minor emissions
Waste transportation	Spills	CO_2, SO_2, NO_x, dust, odour, noise, spills	Spills		Significant contribution of CO_2

CO_2 = carbon dioxide; CH_4 = methane; VOCs = volatile organic compounds; SO_2 = sulphur dioxide; NO_x = nitrogen oxides; N_2O = nitrous oxide; HCl = hydrochloric acid; HF = hydrofluoric acid, CO = carbon monoxide; and PAHs = polycyclic aromatic hydrocarbons.
[a] Assuming no energy recovery.

Table 2
Emissions of greenhouse gases in the EU-15 and solid waste management contribution in 1994 (Source: Smith et al., 2001).

Greenhouse gas	Emissions (Mt)	Global warming potential (GWP) over 100 years	Emissions as CO_2 equivalent (Mt)	% From solid waste disposal
CO_2	3215	1	3215	<0.5
CH_4	22	21	460	33
N_2O	1.05	310	325	1

EU-15: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, Sweden, and UK.

In the case of exposure to ionising radiation, natural background exposure (around 2–3 mSv per year) is the main source for the general population (Sutherland, 2003; Watson et al., 2005). In some parts of the world, due to high levels of uranium and thorium in the local bedrock, the background exposure can be a lot higher, especially when radon gas accumulates indoor or inside mines (up to a few thousands of mSv per year). The maximum recommended occupational limit is 100 mSv (ICRP, 1990) over a period of 5 years (i.e. an average of 20 mSv per year) and no more than 50 mSv in any one of these five years.

The average dose received by nuclear radiation workers is about 2 mSv (UNSCEAR, 2000) though some radiation workers can receive a few tens of mSv. The recommended exposure limit from industrial sources is 1 mSv for the general public. The additional dose received by people residing near nuclear power plants is 0.002 mSv, one order of magnitude less than for those living near coal fired power plants (UNSCEAR, 2000; Sutherland, 2003). Overall, 75% of the average annual dose of ionising radiation is of natural origin, and nearly 25% is due to medical sources, with 0.01% caused by exposure to radioactive material from the nuclear fuel cycle (UNSCEAR, 2000). Records of exposure of miners and resident population of mining districts to ionising radiation and radon gas are very poor. In many countries where uranium deposits are located, regulations and legislation on human health protection are inexistent or inadequate.

Indirect health effects due to the contribution of greenhouse gases from waste disposal activities could be significant. Rising temperatures (and low level ozone levels) due to climate change would affect old people with cardiovascular problems and both old and young people with respiratory problems such as asthma. On the positive side, fewer people would die from cold-related illnesses. Diseases (e.g. malaria) that are spread by vectors such as mosquitoes could become more common. Rising sea levels, flooding and extreme weather events are also likely to cause destruction and casualties. The main cause of global warming is the increasing amount of greenhouse gases (CO_2, CH_4 and N_2O) in the atmosphere. A significant contribution comes from waste management practices (Smith et al., 2001), as indicated in Table 2 which refers to the EU emissions in 1994. The main contribution to the greenhouse effect in the EU is from methane released from landfills where biodegradable waste undergoes anaerobic decomposition. Given the high proportion of waste traditionally landfilled in Europe, the Landfill Directive 1999/31/EC introduced targets for the reduction of biodegradable waste disposed of in landfills in member states, in addition to requirements for the collection of landfill gas. A variety of voluntary and regulatory actions have also been proposed or undertaken in other countries. According to Smith et al. (2001), the estimated overall positive greenhouse gases flux in the EU in 2000 was 50 kg of CO_2 equivalent per tonne of waste. The calculation was based on the average European Union MSW composition and took into account gas emissions, carbon sequestration, and the avoided emissions due to electricity production with landfill gas. Also, if the EU landfill directive target to reduce the landfilling of untreated wastes in 2016 to 35% of 1995 levels is achieved, the estimate for 2020 is a negative flux of about 200 kg of CO_2 equivalent per tonne of waste. Even larger negative fluxes were estimated assuming different scenarios (e.g. more recycling, more incineration with energy recovery, more biological treatment).

5. Epidemiological investigations

Epidemiological studies dealing with the impact of waste management activities on human health are normally *observational,* as opposed to *experimental,* due to ethical reasons. Experimental studies are more typical of clinical trials carried out by/for the pharmaceutical industry, involving a test population (exposed to a specific substance, or drug) and a control (not exposed) population. In this case, the expected outcome is normally a positive one (e.g. good health outcomes as a result of administration of vitamins, reduction of high blood pressure with hypotensive drugs).

There is a large variety of observational studies, and the interested reader can find detailed information in many publications

(e.g. Bertollini et al., 1996; Elliott et al., 2000; Rushton and Elliott, 2003). The most common types of investigations are the following:

(i) *Prospective cohort studies*: Two cohorts of people (the exposed group and the non-exposed group) are assessed over a long period of time during which the degree of exposure of the population and the rate of development of a disease are recorded, in addition to other information collected via questionnaires. These studies normally involve the collection and analysis of human fluids or tissue (e.g. blood, urine, hair, teeth). In order to be able to control the possible confounding factors and to ensure the statistical significance of the results, a large population needs to be enrolled and the overall cost can be considerable.

(ii) *Retrospective case-control studies*: In this case, a case group of people is selected (i.e. patients that have already developed a specific disease), and a control group of healthy people. All participants are interviewed, and information on past exposure is collected retrospectively. These studies are usually less expensive than prospective cohort studies, involve smaller groups of people, and require fewer investigators, but are more prone to bias.

(iii) *Cross-sectional studies*: They are conducted on a specific subgroup of the exposed population over a relatively short period of time. They are the opposite of *longitudinal studies* that are carried out over long periods of time. Cross-sectional studies can be useful to generate hypotheses that can be tested later by more comprehensive studies. They can be effective if the illness investigated is quite a common one, and they are normally cheaper to carry out. Unfortunately, it can be difficult to distinguish whether a particular illness developed before or after the group was exposed to a potential hazard.

Human exposure to substances released at waste management facilities can be (i) *acute* in case of a serious accident causing short-term exposure to high levels of potentially hazardous substances, ionising radiation, bioaerosols, dusts and (ii) *chronic*, when it involves long-term exposure to low concentrations of these substances or radiation.

In most cases, environmental epidemiologists need to investigate the occurrence of clinical effects in a population that may have been affected by emissions slightly above natural background levels. Their task becomes particularly difficult at sites where sanitary landfills, incinerators, or other waste management facilities are state-of-the-art, built with the best available technology, and are operated according to guidelines and in full compliance with legislation. In order to have a reasonable chance of detecting significant clinical differences between a control population and a 'test' population, the investigation needs to have adequate *statistical power* in order to avoid making false positive or false negative conclusions. As the difference in the incidence of specific clinical effects between the two populations is usually small, the power of the investigation relies heavily on sample size. This would normally mean studying at least thousands or tens of thousands of persons in the exposure area and in the control area. The theoretical population size required for a statistically valid interpretation of the data may be larger than the entire population in the geographical area that needs to be studied. The resources required for such investigations are rarely available. A compromise approach is based on a *meta-analysis* that combines the results of a number of single-site studies, though this type of analysis has its own weaknesses (e.g. the difficulty of controlling bias in the original studies, the difficulty accessing studies that showed no statistically significant results and thus remained unpublished).

Other potential limitations in epidemiological studies include: insufficient data on emissions, no data on direct exposure to emissions from waste management facilities and other sources, confounding factors (e.g. ethnicity, income, hazards from other sources), population mobility, and long latency period of some illnesses. These issues are well known in the field of clinical trials, and a number of useful suggestions and guidelines have been published (e.g. Freiman et al, 1978; Moher et al., 2001; Schultz and Grimes, 2005). Schultz and Grimes (2005) also stressed the importance of avoiding "publication bias" and also bias due to the lack of adequate randomisation in the study. Publication bias is common in case of low-powered studies. This means that the literature may include preferentially publications with statistically significant results.

An important step in an epidemiological study is to define the strength of the association between exposure to a potentially toxic substance and specific health effects. This is usually achieved by calculating the ratio of the incidence of a disease in the exposed population over the incidence of the same disease in the non-exposed population. This is called the Relative Risk (RR). In case-control studies, a similar index of association called Odd Risk (OR) is used. An increased risk of developing a specific health outcome would be indicated by an RR > 1. For example, if the RR = 5, the risk is five times higher (or an increase of 400%). Though an increased risk (RR > 1) of a certain disease is found for a specific exposure situation, the actual cause of the health effect may be one that has not been investigated. To judge whether an agent actually causes a health effect, many other issues need to be considered. One of these is the statistical significance of the association found, as one must try and exclude that the association may be due to chance. The confidence level (confidence interval) reported is usually 95%, occasionally 99%. Table 3 shows the model used by the World Cancer Research Fund and the American Institute for Cancer Research (WCRF and AICR, 1997) to define the strength of evidence of an association between exposure and illness. The risk level allocated derives from the combined assessment of the RR and statistical significance found.

Strong evidence of adverse health effects associated with some environmental factors is shown by high RR values (e.g. Tomatis, 1990), but the overwhelming majority of epidemiological studies on health outcomes possibly associated with waste management activities report RR or OR values of less than 1.5, and rarely of more than 2, as shown in Table 4.

An attempt to classify the strength of the scientific evidence was published by SWPHO (2002) and Saffron et al. (2003). It was based on the model used by the World Research Fund to evaluate the role of food and nutrition in the prevention of Cancer (WCRF and AICR, 1997). A total of 1035 publications published since 1982 in Europe, the USA, Australia and New Zealand were reviewed. Health risks from ionising radiation were not included. Details of the algorithm used in the appraisal of the epidemiological evidence can be found in Saffron et al. (2003). Their main conclusion of the overall assessment of the literature was that the evidence of adverse health outcomes for the general population living near landfill sites, incinerators, and composting facilities is 'insufficient'. 'Convincing' evidence was found to exist for

Table 3
Simplified version of the relative risk (RR) and odd ratio (OR) model used by WCRF and AICR (1997).

RR or OR	Statistical significance	Strength of evidence
0.87–1.5	No	No association
1.5–2.0	No	No association
1.5–2.0	Yes	Moderate
>2	No	Moderate
>2	Yes	Strong

Table 4
Examples of RR or OD values for the association between environmental factors and health, and between exposure to waste management facilities and health.

Cause	Effect	RR, OR	CI
Environmental factors (Tomatis, 1990)			
Hepatitis B virus	Liver cancer	104	51–212
Cigarette smoking	Lung cancer	7.8	na
1–14 Cigarettes	Lung cancer	12.7	na
15–24 Cigarettes	Lung cancer	25.1	na
⩾25 Cigarettes			
Landfilling			
Nant-y-Gwiddon, UK (Fielder et al., 2000)	Congenital anomalies		
Before site opened,1983–1987	1.9	1.3–2.9	
First 2 years of operation,1988–1989	3.6	2.3–5.7	
From 1990 to 1996	1.9	1.2–3.0	
9565 UK landfill sites (Elliott et al., 2001a,b)	Congenital anomalies	1.01	1.01–1.02
Low birth weight	1.04	1.03–1.05	
Cardiovascular defects	0.96	0.93–0.99	
Incineration			
Besançon incinerator, France (Viel et al., 2000)	Non-Hodgkin's lymphomas	1.27	1.1–1.4
	Soft tissue sarcoma	1.44	1.1–1.9
13 Incinerators, France (Viel et al., 2008)	Non-Hodgkin's lymphomas	1.12	1.002–1.251
Incinerator and other industrial sources of dioxins, Italy (Zambon et al., 2007)	Sarcoma: population with the highest exposure	3.30	1.24–8.76
Sewage contaminated water			
Sydney beaches, Australia (Kay et al., 1994)	Gastroenteritis, from exposure to:		
0–39 Faecal streptococci/100 mL	1.0		
40–59 Faecal streptococci/100 mL	1.91	1.60–2.28	
60–79 Faecal streptococci/100 mL	2.90	1.43–5.88	
80+ Faecal streptococci/100 mL	3.17	1.12–8.97	
Sydney beaches, Australia (Corbett et al., 1993)	Any symptom[a], from exposure to:		
10–300 cfu/100 mL	2.9	1.7–5.1	
300–1000 cfu/100 mL	3.8	2.1–7.1	
1000–3000 cfu/100 mL	5.2	1.7–16.0	
Exposure to ionising radiation			
Nuclear industry workers (Cardis et al., 2005)	All cancers excluding leukemia	0.97[b]	0.14–1.97
	Leukemia, excluding chronic	1.10[c]	
	Lymphocytic leukemia	1.93[b]	<0–8.47
		1.19[c]	

na = Not available.
[a] Cough, ear or eye symptoms, fever, gastrointestinal symptoms; cfu = colony-forming units.
[b] For a radiation dose of 1 Sv.
[c] For a radiation dose of 100 mSv.

gastrointestinal problems associated with ingestion of sewage-contaminated recreational waters, while the evidence was "probable" for airways symptoms in workers of sewage treatment plants.

Though based on different criteria, a number of relatively recent reviews on this topic reached similar conclusions for the human health impact of the waste management practices that they reviewed. Examples include Bunger et al. (2000), IEH (2000), Vrijheid (2000), WHO (2000, 2007), Environment Agency (2001), Hu and Shy (2001), Redfearn and Roberts (2002), Rushton (2003), DEFRA (2004), Franchini et al. (2004), Harrison (2007), and Sykes et al. (2007).

6. Health effects of waste management activities

6.1. Landfilling

Many reviews suggested an association between proximity or exposure to landfill sites and ill health. For example, Sever (1997) and Johnson (1997, 1999) highlighted an increased risk of birth defects and some cancers for the population living near landfill sites. Vrijheid's (2000) review of the literature published over almost 20 years (1980–1998) indicated similar associations, but most studies lacked data on direct exposure. The scientific evidence of adverse health effects from landfills was described as poor or inadequate by Redfearn and Roberts (2002) and Wigle (2003). The World Health Organisation published the findings of two workshops, the first on the health effects of waste landfills (WHO, 2000),

and the most recent (WHO, 2007) on the health effects of landfills and incineration. In both cases, the main conclusion is that the evidence that links waste landfills and incinerators to health endpoints (especially cancer, reproductive outcomes and mortality) is either inadequate or insufficient. Two large multi-site studies carried out in the UK are worth mentioning. Elliott et al. (2001a,b) included about 80% of the UK population residing within 2 km of 9565 landfill sites (7803 non-special, 774 special, 988 unclassified) between 1983 and 1998. Given the influence of confounding factors and data artefacts, no clear cause could explain the slight excess risk of birth defects and low birth weights that was found. The study of Jarup et al. (2007) found no excess risk of giving birth to a child with Downs syndrome in the population residing near 6289 landfill sites in England and Wales. Some of reports that have shown high relative risks (RR) for congenital anomalies did not take into account important confounding factors. In the case of the Nant-y-Gwiddon landfill site in the UK (Table 4), Fielder et al. (2000) failed to take into account that the local residents had been also exposed to the emissions from highly polluting local incinerators well before the landfill operation started (Roberts et al., 2000).

6.2. Incineration

Though incinerators can potentially emit a number of pollutants (see Table 1), the main concern about incinerators has been the emission of a group of persistent organic compounds known as "dioxins", more specifically polychlorinated dibenzo-*p*-dioxins

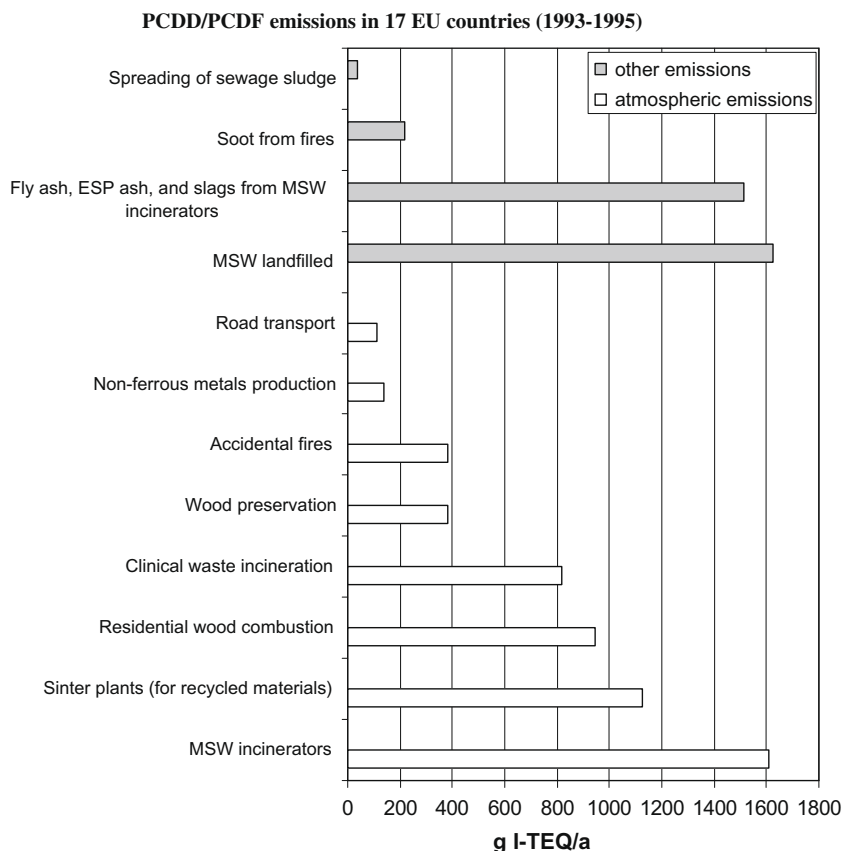**PCDD/PCDF emissions in 17 EU countries (1993-1995)**

Fig. 5. Main emissions of PCDDs/PCDFs in 17 EU countries in 1993–1995. Source of data: Quaβ et al., 2000.

(PCDDs), polychlorinated dibenzofurans (PCDFs), and polychlorinated biphenils (PCBs). PCDDs and PCDFs are produced by combustion processes, and mostly by incomplete combustion of municipal waste, medical waste, household waste, by forest fires, by burning wood and coal, during the manufacture of pesticides and other chemicals, and are present in tobacco smoke and car exhaust (Dyke et al., 1997; Fielder, 2007). These substances are quite resistant to biodegradation, they accumulate in food (dairy products, eggs, fish, animal fat), and many (29) are considered to be toxic (USEPA, 1994a, 1994b). Dioxin-like toxicity is attributed to 7 of the 75 possible PCDD congeners, to 10 of the 135 possible PCDF congeners, and to 12 of the 209 possible PCB (USEPA, 1994a, 1994b). The most toxic (carcinogenic), according to the International Agency for Research on Cancer (IARC, 1997), is 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), based on laboratory experiments on animals and on cohort studies of groups living in industrial areas.

The toxicity of dioxins is indicated in Toxicity Equivalent Factor (TEF) units. The TEF of TCDD (the most toxic) is rated as 1, and the others less than 1. On the assumption that the effect of various dioxins is additive, the TEF value of each dioxin is multiplied by its concentration and a final toxic equivalent quantity is obtained, the International Toxic Equivalent Quantity (I-TEQ). Quaβ et al. (2000) produced the first inventory of 1993-1995 industrial and non-industrial emission sources of PCDDs and PCDFs in 17 European countries (EU 15 plus Sweden and Switzerland). Despite some uncertainties due either to lack of data or to the poor quality of some of the original data included in the study, this work shows the complications faced by health impact investigators due to the wide range of potential sources, and some of the possible causes of confounding factors in epidemiological studies. A summary of the main sources of PCDDs and PCDFs in 1993–1995 is shown in Fig. 5. The main source of PCDDs/PCDFs emis-

sions to air was MSW incineration, and clinical waste incineration was also a large contributor. However, sinter plants for recycled materials and residential wood combustion represented about 38% of the total emissions. Since the EU set an emission limit (0.1 ng/m^3 I-TEQ) for incineration plants, pollution abatement technologies have reduced emissions from MSW incinerators significantly, and in some EU countries all incinerators comply with this limit. If all EU incinerators complied with this requirement, the total annual atmospheric emissions would be about 20 g I-TEQ (Quaβ et al., 2000).

The diseases often investigated in epidemiological studies around MSW incinerators are the non-Hodgkin's lymphoma and soft tissue sarcomas. Many studies have been carried out, particularly in France, the country with the largest number of incinerators in the EU. Examples are the investigation around the Besançon incinerator (Viel et al., 2000), and around 13 other French incinerators (Viel et al. 2008). These facilities were responsible for high emissions of dioxins. Though a link between non-Hodgkin's lymphoma and exposure (based on residence and soil levels of dioxins) to dioxins emitted by municipal solid waste incinerators was found, the strength of this association was weak, as shown in Table 3. Also, Viel et al (2008) stated that these results cannot be extrapolated to modern low-emissions incinerators and that other pollutants (e.g. polycyclic aromatic hydrocarbons, heavy metals) emitted by incinerators may also have been responsible for cancers in the exposed population.

In a case study carried out by Zambon et al. (2007), exposure to dioxin-like substances in the Province of Venice was shown to be responsible for an increased risk of developing sarcoma. In the study region, there were 33 sources of airborne dioxin, including incinerators of municipal solid waste, industrial waste, and medical waste, and a range of other industrial sources including an oil

refinery. Emission levels of dioxins (the peak period being 1972–1986) and exposure levels were not known and had to be recalculated and modelled. No information on levels of dioxins in the food that is known to accumulate dioxins was available for much of the period investigated.

Despite the best effort of epidemiologists to reconstruct emissions of pollutants from incinerators that have closed down and to estimate exposure by means of residence records or of atmospheric models, there is still a degree of uncertainty about the actual exposure of resident populations (e.g. Franchini et al., 2004). Though waste management facilities were not involved, a good example of useful exposure measurements is reported in the work of Bertazzi et al. (2001) on the population of Seveso (Italy), famous for the industrial accident of 1976. These measurements indicated an RR of 2.8 (95% confidence interval = 1.1–7.0) of developing non-Hodgkin's lymphoma. In this extreme case, the mean TCDD level (136 ng/kg) in the blood lipids was about 50 times higher than typical background levels.

In the USA, the National Research Council (NRC, 2000b) came to the conclusion that epidemiological studies could not detect any significant excess health effects. Though most studies reviewed by Hu and Shy (2001) showed higher concentrations of heavy metals and organic chemicals in the populations residing closer to incinerators, a causal link could not be proven. A review of the Department of Environment Food and Rural Affairs (DEFRA, 2004) included a total of 102 publications and concluded that there is no convincing evidence of a link between incineration and cancer or respiratory problems, and between landfilling and cancer.

It is also important to mention that the main exposure to dioxin-like substances is via food (seafood, dairy products, animal fats and eggs) contaminated with PCDDs and PCDFs (Kishimoto et al., 2001).

6.3. Sewage treatment and landspreading

A number of papers (Pruss, 1998; IEH, 2000) focussed on the health effects of bathing water polluted with bacteria derived from sewage released from sewage treatment plants. They reported convincing evidence of a high risk of increased symptoms (mostly gastrointestinal problems, but also respiratory and ear and eye symptoms) associated with pathogens released into water bodies, though there is much controversy about the real cause of these illnesses (i.e. viruses as opposed to bacteria).

There is insufficient information on the fate of pathogens that are incorporated in the soils after inputs of biosolids, and on the survival of pathogens after they are transferred to surface water or ground water. Animal manure, sewage sludge, and compost (especially compost derived from treatment of manure in windrows) may contain large numbers of pathogens such as *Salmonella*, *Campylobacter*, *Escherichia coli*, *Giardia* and *Cyptosporidium* (e.g. Gerba and Smith, 2005) and viruses. In the UK, the huge increase in reported cases of food poisoning was thought to be due to the presence of *E. coli* O157 in the organic waste applied to agricultural soil (Jones, 1999). Risks to human health arise at the spreading stage (as bioaerosols), but also as a result of bacterial and viral contamination of surface water from runoff (Servais et al., 2007). Freshwater and marine water contamination incidents in different parts of the world have been associated with discharges from sewage treatment plants and as a result of land application of soil amendments, especially manure (e.g. Goss and Richards, 2008).

6.4. Composting

According to most review papers (e.g. Maritato et al., 1992; Bunger et al., 2000; Environment Agency, 2001; Harrison, 2007) the main concern is for compost workers, as they are more likely to de-

velop respiratory and dermal illnesses than the general public. The overwhelming majority of household waste and green waste composted is treated with open-air windrow systems, though there is a gradual shift towards in-vessel plants. Respiratory illnesses can be caused by exposure to dust and bacteria, funghi, actinomycetes, endotoxins and 1-3 β glucans released at composting facilities. Few studies were found on the health impact on residents living close to composting facilities, mostly publications on bioaerosol dispersion from composting sites. Inconclusive evidence of increased ill-health compared to control populations was found (Millner, 1995). The definition of a safe buffer distance is still hampered by the limited knowledge on the hazards associated to bioaerosols, and on dose–response relationships. The review of Domingo and Nadal (2008) is the most recent publication found on the chemical and biological risks for workers of composting facilities and on the potential impact on local residents. Examples of epidemiological studies that indicated an association between bioaerosol pollution derived from outdoor composting facilities and irritative respiratory symptoms in nearby residents include Herr et al. (2004a,b).

6.5. Radioactive waste management

Most studies of occupational exposure to a few mSv in excess of background levels (such as the additional dose to the public from emissions from nuclear power plants) could not link this additional dose of radiation to human illnesses; in the case of chronic exposure to low doses, adverse effects could not be statistically associated with the measured increased levels of exposure (Shihab-Eldin et al., 1992; Cohen, 1995; Sutherland, 2003). In Europe, a number of studies (e.g. Black, 1984; Heasman et al., 1986; Viel et al., 1995; Guizard et al., 2001) were carried out in the UK and France, mostly as a result of reported clusters of leukemia in children residing close to nuclear plants or nuclear reprocessing facilities. These studies did not produce conclusive evidence of a link between residence near nuclear sites and adverse health effects. The results of a case-control study (Gardner et al., 1990) of the incidence of leukemia and lymphoma among young people near the Sellafield nuclear power plant in the UK caused some concern, but the subsequent investigation by COMARE (2002) came to the conclusion that no valid statistical base existed for such cause–effect relationship. The largest retrospective cohort study (Cardis et al., 2005) carried out on 598,068 workers (90% men) in the nuclear industry in 15 countries concluded that there was a small excess risk of cancer for a cumulative dose of 100 mSv (90% of workers received cumulative doses of less than 50 mSv, 5% received more than 100 mSv, and less than 0.1% received cumulative doses of more than 500 mSv), equating to 1–2% of deaths from cancer attributable to radiation. As shown in Table 4, the relative risk found for all cancers excluding leukemia was 1.10, and for leukemia, excluding chronic lymphocytic leukemia, it was 1.19 for a radiation dose of 100 mSv. Reviews of epidemiological studies include those of Laurier and Bard (1999) and Doll (1999).

6.6. Waste management, occupational health and safety considerations

The health and safety performance of the waste management industry is likely to vary significantly across the world, with major differences between developed and developing countries. In developed countries, workers protection and health and safety measures have substantially reduced the likelihood of fatal or major accidents. In the UK, about 160,000 workers are employed in the waste management sector, and a total of about 3800–4300 accidents are reported every year (HSE, 2004). This translates into an accident rate that is about four times the national average; the fatal injury accident rate is about 10 per 100,000 workers (i.e. 10 times the na-

tional average), and the major injury rate is about 330 per 100,000 workers (i.e. more than three times the national rate), with most accidents (handling/sprain injuries, being struck by refuse collection vehicles) occurring during refuse collection and unloading (HSE, 2004).

Epidemiological studies on the occupational health impact of recycling are extremely rare. Given the increased recycling rate in many developed and developing countries, research in this field is urgently required. In developing countries, the main issue is associated with infections and injuries from unregulated recycling (scavenging) in open dumps.

6.7. Waste management, epidemiology and biomarkers

The total amount of a toxicant to which a person is exposed is not necessarily directly correlated with possible adverse effects. It is the amount that becomes bioavailable, and more specifically the amount that is absorbed by a particular organ, tissue, cell, that may cause a toxic effect and a health outcome. And even if this *effective dose* were known, individuals are likely to react differently depending on age, gender, and genetic susceptibility. The presence of toxic substances in groundwater or surface water contaminated by leachate from a badly engineered and operated landfill does not necessarily imply exposure of the resident population, as drinking water in the region may be sourced elsewhere. This highlights the weaknesses of epidemiological investigations that base their dose–response relationship on exposure data derived from emission information from point or diffuse sources or, even worse, on surrogates of exposure such as residence or distance from waste management facilities.

Biomarker epidemiology has developed important new approaches to the assessment of environmental and occupational health that have the potential of overcoming these weaknesses in epidemiological investigations. Biomarkers are biological parameters or indices that can be used to measure at cellular or molecular level the exposure to effective dose of toxic substances and their adverse effects. Biomarkers can be broadly classified into three groups, though there is some overlap between these: (i) exposure biomarkers, (ii) health effect (outcome) biomarkers, and (iii) susceptibility biomarkers.

Exposure biomarkers can be xenobiotic substances or metabolites found in the human body, or substances that are produced from the interaction between xenobiotics or metabolites with other substances present in the body. Their concentrations are usually determined in blood, serum, urine, teeth, and adipose tissue, but breath biomarkers have also been determined (Pleil, 2008). Exposure biomarkers are also cellular, molecular or DNA changes, and inherited gene variations resulting from exposure to toxic substances (Schmidt, 2006; Pleil, 2008). Exposure biomarkers allow an estimate of the amount of toxic agent absorbed by each individual.

Health effect biomarkers are indicators of the abnormal functioning of the body or a specific organ as a result of exposure to a specific substance (specific biomarkers) or to a group of substances (non-specific biomarkers). They allow the assessment of the impact (permanent or reversible) of the absorbed toxicant.

Susceptibility biomarkers are parameters (physical, chemical, genetic) that can make a person more sensitive to a toxic substance and increase the health risks arising from exposure. For example, single-nucleotide polymorphisms (SNPs) are inherited gene variations that can increase or reduce disease susceptibility following environmental exposure (Schmidt, 2006).

Biomarkers allow early detection (at cellular or molecular level) of pathological changes associated with exposure to chemicals and radiation. The usefulness, weaknesses, and the applicability of biomarkers to environmental and occupational epidemiology can be found in many publications (e.g. Gunn et al., 1991; WHO, 1993; Indulski, 1995; Bonassi, 1999; Wild et al., 2002; Vasseur and Cossu-Leguille, 2003; Linzalone and Bianchi, 2009). A few examples of the application of biomarkers in environmental and occupational epidemiology are mentioned here. Unfortunately, a literature search did not provide many examples of the use of biomarkers to assess the effects of exposure of the general public to emissions from municipal solid waste management facilities.

The determination of biomarkers of genotoxic damage (e.g. chromosomal aberrations in lymphocytes) and cell proliferation indices can be used for early detection of an association between exposure to hazardous waste and DNA damage/changes (e.g. Gonsebatt et al., 1995). It is likely to be more difficult to find genotoxic effects in the local resident population, though it may be possible to detect an abnormal DNA repair response compared to cells from a control population. This was shown in a case study (McConnell et al., 1998) of residents exposed to waste from uranium mining and milling. In this study, radiogenic isotopes were used as biomarkers of exposure, i.e. ^{238}U and Pb isotope ratios in soil samples.

Staessen et al. (2001) found that biomarkers of renal disfunction were positively correlated with Pb levels in the blood of adolescents living near a Pb smelter. In the same study, biomarkers of DNA damage were positively correlated with urinary metabolites of polycyclic aromatic hydrocarbons (PAHs) and volatile organic compounds (VOCs). Kap-Soon et al. (2004) used the 1-hydroxypyrene (1-OHP) urinary metabolite to assess exposure to PAHs. The 1-OHP concentration in the exposed groups was 0.28 µmol/mol creatinine, and 0.078 µmol/mol creatinine in the unexposed (control) groups.

Yoshida et al. (2005) measured dioxins and creatinine in serum and urinary estrogen metabolites of 57 male waste incinerator workers and concluded that creatinine increased with increasing dioxins levels, and that dioxins metabolise estrogens to 16-hydroxyestrogens.

In the study of Yoshida et al. (2003), blood and urinary levels of oxidative stress markers were measured in 81 municipal solid waste incinerator workers. The concentration of urinary 8-OH-dG increased significantly in workers exposed to fly ash, which is known to contain PCDDs, PCDFs, PAHs and heavy metals.

Existing environmental surveillance standards can be inadequate because biological effects can be detected only when illnesses are diagnosed. Biomarkers allow investigators to detect excessive exposure and changes at cellular or molecular level before a potential outcome reaches the clinical stage. Therefore, timely and adequate preventive action to reduce possible risks becomes possible.

7. Conclusions

The existing epidemiological evidence linking waste management and human health is quite controversial. Most studies investigated health impacts of old types of waste management facilities, especially in the case of incinerators. There is very little data on direct human exposure, and most studies resorted to surrogates such as residence information, with most recent studies including data on potential exposure pathways (e.g. pollutant concentration in soil, modelled atmospheric exposure). Confounding factors have not been adequately controlled in many studies, especially social deprivation and exposure to other sources rather than the one investigated.

In the case of landfills, the strongest association with human health is for congenital malformations. Incineration is often reported to be associated with an increased risk of developing non-Hodgkin's lymphomas and sarcomas. The dose–response of serum dioxins suggests that the main intake pathway of these substances is not inhalation but food. Few studies are available on new-generation incinerators fitted with modern emission-abating technol-

ogy, and any future epidemiological investigations will find it quite difficult to detect excess adverse effects as these will become even more difficult to measure.

There are very few studies on the health impact of composting on resident populations, but there is some evidence that compost workers have significantly more diseases of the respiratory tract and increased antibody concentrations against funghi and actinomycetes. As the spreading of soil amendments (including sewage sludge and manure) has increased considerably in many countries, there is a need for more research into the effect of potential pathogens (via bioaerosols, via food contamination, via soil erosion and mobilisation into water bodies) on human health. Most studies on landspreading are about occupational diseases, the remaining studies are on respiratory illnesses and gastrointestinal symptoms associated with contaminated bathing waters. There is convincing evidence of a high risk of increased symptoms associated with pathogens originating at sewage treatment plants. In most cases, a significant dose–response relationship has been shown, especially with enterococchi and faecal streptococchi. Given the increasing evidence of the role of viruses as the cause of human infections in sewage-contaminated waters, the lack of research in this area is surprising, though this is likely to be due mostly to the costs involved in viral investigations.

High level radioactive waste produced by nuclear facilities in the form of spent nuclear fuel is accumulating in many countries while a decision is taken about a final repository. The construction of many new nuclear power stations in various countries, especially in Asia, is likely to increase the volume of radioactive waste to be disposed of. At the same time, decommissioning of old nuclear power stations will produce large volumes of low level radioactive waste. The risk of excess exposure to ionising radiation is a very emotive issue in areas where nuclear reactors, spent fuel storage facilities and nuclear waste reprocessing facilities are in operation or are at the planning stage. If the best technologies and controls are used in the construction of new reactors, in the decommissioning of old ones, and in the operations associated with fuel enrichment and recycling facilities, radioactive waste dispersion in the environment and exposure of the general public to ionising radiation should be minimal. Most epidemiological studies did not produce conclusive evidence of a link between residence near nuclear sites and adverse health effects. However, public health surveillance is of paramount importance. Public health protection needs to include new molecular monitoring techniques for the early identification of exposure to ionising radiation so that preventive action can be taken before illnesses develop. In order to reduce the uncertainties in epidemiological research it is necessary to focus on prospective cohort studies of sufficient statistical power, consider confounding factors and bias (including publication bias), to select biomarkers that are specific to the health risk investigated, and to have access to detailed records of environmental exposure (air, water, soil, food) to toxicants, and to the details of the characteristics and operation of waste management facilities.

In many developing countries, lack of resources and political will, poor education and widespread illnesses due to bad sanitation and potable water, make waste management a low priority. Poor communities often obtain their livelihood from salvaging solid waste for recycling and are affected by parasites and intestinal infections, not to mention injuries from sorting solid waste. In developed countries, public concern about the location of landfills and incinerators is largely based on the effects on human health of notorious cases of poor management of industrial waste (e.g. Love Canal in the USA in the 1970s) and landfill sites (Loscoe, UK, 1986; Naples, Italy, 2008), or more commonly, to major industrial accidents unrelated to waste management operations, such as Seveso (Italy) in 1979 and Bhopal (India) in 1984, but there is also fear about possible adverse effects in the general population residing near relatively modern landfills and incinerators. However, the overwhelming majority of epidemiological studies have not managed to prove convincingly and unequivocally that excess risk of contracting specific illnesses is associated with waste management facilities. This is due to the limitations of environmental epidemiological studies and to improved technology and organisation of waste management activities. There is a need to set up well-designed epidemiological studies capable of giving evidence of the effect of exposure to low levels of potentially hazardous substances. It is extremely important to have direct human exposure data, especially from exposure biomarkers, possibly collected before (not only during and after) a waste management facility becomes operational.

The level of significance of the risks to develop cancers or other illnesses from emissions from waste management facilities should also be seen in the overall context of other risks to the local population, not only with reference to the appalling mortality rates in developing countries due to lack of safe drinking water, poor sanitation, lack of sewage treatment plants and of adequate waste disposal facilities, but also with the typical health risks recorded in affluent societies. Last but not least, it is important to consider the benefits to public health that derive from disposing waste in state-of-the art facilities, until ways are found to minimise, re-use, and re-cycle more waste.

On a global scale, given the accelerated industrialisation and urbanisation in developing countries, billion of tonnes of waste are produced every year. The health issues associated with the disposal of waste are escalating in countries such as China and India, to name a few. Massive investment in waste management facilities, training and education is required in order to reduce the health impact of inappropriate waste disposal methods. However, the issue of health risks associated with waste management also needs to be tackled on many other fronts, i.e. (i) introduction of measures and incentives for waste minimisation, waste prevention, recycling, and composting, (ii) addition of waste management costs to consumer products, (iii) more public participation in the choice of waste management practices at local and regional level, (iv) public health surveillance, and (v) the use of biomarker epidemiology techniques in future investigations.

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