ENVIRONMENTAL HEALTH

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Heavy Metal and PAH
Compounds from Municipal
Incinerators

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EMISSIONS OF HEAVY METAL AND PAH COMPOUNDS FROM MUNICIPAL SOLID WASTE INCINERATORS: CONTROL TECHNOLOGY AND HEALTH EFFECTS

Report on a WHO meeting

Florence
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FOREWORD

In recent years, the incineration of municipal solid waste and the resultant emission of toxic substances have generated worldwide concern about possible adverse health effects. Such concern in Italy prompted two meetings organized by the WHO Regional Office for Europe at the request of Italian regional authorities in 1986 and 1987. The first meeting focused on the emissions of dioxins and furans and the second meeting on the emissions of PAH and heavy metal compounds.

This document reports on the deliberations, conclusions, and recommendations of the second meeting held in October 1987 in Florence to discuss control technology for and health effects from incinerator emissions of heavy metal and PAH compounds. The meeting was initiated and financed by the Regional Government of Tuscany, in collaboration with the Government of Italy.

This report is aimed at the public and environmental health authorities of the Tuscany Region, but it should be equally useful to such authorities elsewhere as well as to professonals in this area.

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Regional Officer for Environmental
Health Hazards

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INTRODUCTION

Emissions consisting mainly of dust, smoke, heavy metal compounds, and noxious gases from municipal solid waste (MSW) incinerators, sewage sludge incinerators, hazardous waste incinerators, heating facilities, and thermal power plants have long aroused concern. More recently, the potential danger of hazardous organic compounds that may be present in emissions, such as polycyclic aromatic hydrocarbons (PAH), has been recognized. The high toxicity of some of these compounds has called into question the safety of incinerating MSW. Answering these questions requires a better understanding of the emission levels and potential adverse public health effects of heavy metal and organic compounds, such as PAH, emitted from MSW incinerators.

The WHO Regional Office for Europe organized a Working Group at the request of the Regional Government of Tuscany, in collaboration with the Government of Italy. The purpose of the meeting was to review and discuss data and public health aspects related to the emission of heavy metal and PAH compounds during incineration of MSW. The Working Group's objective was to give national and local health authorities, as well as other interested parties, guidance on the public health significance of these emissions and, when possible or necessary, to point out practicable approaches to minimize the emissions and the related public health impacts. The meeting was attended by 26 participants from 13 countries (Annex 2).

Dr M. Piscator was elected Chairman, Mr A. Finkelstein and Dr K. Kostial were elected Vice-Chairmen, and Mr C.O. Velzy and Dr A. Wadge each served as Co-Rapporteur. Dr M.J. Suess acted as Scientific Secretary.

This meeting is the second on municipal waste incinerators organized by the WHO Regional Office for Europe in Italy. A working group on dioxin and furan emissions met in Naples in 1986. The report on the first meeting is available from the WHO Regional Office for Europe [1].

The Working Group focused on the following major items:

- influence of the composition of the material being incinerated, the conditions of the incineration process (temperature, mixing, retention time of combustion products, oxygen, and the operational aspects), and pollution control technology and its operation on the quality and quantity of emissions;

- environmental pathways of emitted heavy metal and PAH compounds as related to human exposure;

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- relative importance of MSW incineration compared with other sources of human exposure to heavy metal and PAH compounds;

- public health significance of emissions of heavy metal and PAH compounds during MSW incineration.

The Working Group broke into two subgroups (Annex 1). Subgroup 1 reviewed available data on emissions of heavy metal and PAH compounds from MSW incinerators. This subgroup also considered and evaluated aspects of design and operation to minimize incinerator emissions. The results of these analyses and considerations were organized into tables of expected emission concentrations from incinerators of different types and with different operational characteristics. Much of the descriptive text material on waste and incineration in this report was extracted from the WHO Naples meeting report [1].

Subgroup 2 summarized the potential human health aspects of exposure to cadmium, mercury, lead, and PAH. It then evaluated the relative contribution that emissions from incinerators may make to the overall exposure of the general population to these pollutants. As a starting point, subgroup 2 compared estimates of ground level concentrations of emissions from MSW incinerators, developed from measured stack emissions of these pollutants, with observed ambient concentrations when available. Pollutant emissions were also compared with inventories of regional or national emissions when available. These comparisons indicated different degrees of importance for each pollutant of concern.

WASTE AND INCINERATION

MSW is a diverse and heterogeneous material generally containing domestic and commercial waste. Domestic waste is solid waste produced in the normal course of household activity or waste of a similar character from institutions. Commercial waste is waste from shops and offices, usually collected with domestic waste. MSW can vary widely in physical composition (e.g. form, size, and consistency) and chemical composition (e.g. carbon, hydrogen, oxygen, nitrogen, chlorine, sulfur, metals, pesticides, plastics, and halogenated hydrocarbons). Industrial waste can introduce substantial quantities of contaminants into the solid waste stream that would otherwise exist only in trace amounts in MSW collected from residential and commercial areas. Only domestic and commercial solid wastes are considered in this report.

A number of options exist for the processing and disposal of MSW. In the past, some countries disposed of these materials at sea. This method is now generally considered to be unsatisfactory because of obvious adverse effects on, for example, marine life, and less

obvious effects, such as the introduction of contaminants into the food chain. Alternate means of MSW disposal include deposition in carefully designed, constructed, and operated landfills, composting or separating component materials for recycling or reuse, or incineration. All of these disposal options produce residuals that must ultimately be disposed of in properly managed landfills. In this report, only incineration is considered from a detailed processing standpoint.

Incineration is defined in the WHO Glossary on solid waste [2] as "the process of reducing combustible waste to an inert residue by high temperature burning". This process has several outputs, such as solid, liquid, and gaseous emissions, and heat energy. The primary function of an incinerator is to reduce waste to an inert residue with minimum adverse impact on the environment. Thermal efficiency, while important, is secondary to this function. MSW incinerators are usually classified according to their capacity, the nature of the waste to be burned, the type of grate system used, and the method of waste heat extraction, flue gas cooling, and gas cleaning. The capacity of an incinerator in terms of quantity of waste processed per unit of time is roughly inversely proportional to the heat value of the waste.

Modern incineration plants reflect significant advances made in solving the technical and practical difficulties of material handling and combustion control. In the early days of waste incineration when air pollution regulations were undemanding or nonexistent, relatively simple, fixed-grate plants operating on a single- or two-shift basis were common. With increasingly stringent regulations to control air pollution, more complex plants requiring continuous operation have become the norm. Frequently, regulations and/or requirements make small plants prohibitively expensive to operate.

In this report, the removal efficiencies and emission test results from different installations are presented in terms of "traditional" and "advanced" incinerators. Traditional incinerators include all facilities incorporating normal combustion controls plus dry particulate collection devices with an efficiency of approximately 95 to 97%. Advanced plants include facilities incorporating modern combustion control systems and acid gas control scrubbing equipment with a particulate control efficiency of 99%+.

Types and characteristics of waste

Two types of waste are considered in this report: as-collected or as-received MSW and processed MSW.

As-received MSW

Some typical compositions of as-collected MSW can be calculated (Table 1). In western Europe, waste composition varies over a wide

range. A typical analysis of MSW samples in western Europe and United States is shown in Table 2.

Recent tests on MSW at two plants in the United States conducted by the US Bureau of Standards [5] have shown average chlorine contents ranging from 0.45 to 0.89% by mass of the total dry as-received waste. Of the total chlorine, approximately 0.14 to 0.46% by mass was from plastic materials, while approximately 0.31 to 0.43% by mass was from all other material. Thus, the variation in chlorine from plastics was about 2.3 times the minimum level at 0.14% by mass, while the variation in chlorine from all other material was about 0.4 times the minimum level at 0.31% by mass. Therefore, the chlorine content in MSW probably depends greatly on the total plastic content.

Processed MSW

Waste processing or separation technologies can affect the chemical composition of MSW, which can vary widely and include or-

Table 1. *Average composition of as-collected MSW*

Type of waste	Western Europe		USA	Middle East city
	%[a]	Range in %[b]	%[a]	%[a]
Organic	21.3	15–40	22.6	62.3
Paper	27.4	8–43	45.6	25.3
Textiles	3.5	1– 6	4.5	1.4
Plastics	3.1	3– 7	2.6	5.8
Glass	9.5	5–11	6.2	1.0
Metals	8.5	3– 8	9.1	2.8
Ash dust	19.8	2–60	7.6	NA
Miscellaneous	6.9	4	1.8	1.4
Total	100.0		100.0	100.0

[a] [3].

[b] Personal communication from experts at the WHO meeting on PCDD and PCDF Emissions from Incinerators for Municipal Sewage Sludge and Solid Waste – Evaluation of Human Exposure, Naples, 17–21 March 1986.

NA = Not available.

Table 2. *Analysis of MSW*

Type of analysis	Weight by %[a]	
	Western Europe[b]	United States[c]
Proximate analysis		
Combustible	42.1	50.3
Water	31.0	25.2
Ash and inert material	26.9	24.5
Total	100.0	100.0
Ultimate (elemental) analysis of combustibles		
Carbon	51.1	50.9
Hydrogen	7.1	6.8
Oxygen	40.1	40.3
Nitrogen	1.2	1.0
Sulfur	0.5	0.4
Chlorine	NA	0.6
Total	100.0	100.0

[a] Gross heat value, as fired = 9000 kJ/kg.
[b] [3].
[c] [4]; Domalski, E.S. et al., paper given at the 1986 National Waste Processing Conference.
NA = Not available.

ganic and inorganic compounds. MSW can contain the following inorganic compounds: cadmium, as a pigment, a stabilizer in plastics or a surface coating, or in alloys and batteries; lead, in batteries, lead-based paints, various metal items, and dusts; mercury, in batteries, and in electrical and control equipment, and organomercuric compounds in paper manufacturing; beryllium, in fluorescent light bulbs; and inorganic metallic salts, in paper manufacturing [6]. Some organic compounds can be potentially troublesome: hexane, toluene, and xylene in oil and acrylic paints, turpentines and other solvents, and lithographic inks; methylene chloride, in paint strippers and propellents for spray paint aerosol cans; formal-

dehyde, in particle board and plywood; dioxins, a contaminant in pentachlorphenol wood preservatives; and various organic compounds, in paper manufacturing.

Some of these substances, such as lead, mercury, and cadmium, can contaminate the environment, regardless of the MSW disposal method. Therefore, when developing a management plan for the disposal of MSW, consideration should be given to collecting separately some material containing these compounds, such as batteries, and disposing of this material under tightly controlled conditions. In other cases, consideration should be given to substituting less hazardous materials. This approach can best be achieved at a national or international level.

Trace organic compounds are widely distributed in the total mass of MSW, and most of them can be destroyed in a properly designed and operated waste combustion unit. However, some types of waste processing, such as composting or materials recovery (e.g. reclamation of paper fibre) may not destroy these substances [6].

Many countries have voluntary programmes of source separation, including Italy, the United Kingdom, and the USA. Such programmes have not had a significant impact on the composition of MSW. In areas where bottles are returned and reused, as much as 50% of the glass may be recycled [7]. However, as glass usually represents 10% or less of MSW, the combustion process will be little affected. Paper collection at the source is usually restricted to newspapers and magazines. In countries with advanced separate collection systems, up to 45% of the paper consumed may be recycled. This reduction may considerably lower the heat value of MSW.

Several programmes have been initiated to remove certain potentially toxic substances from the solid waste stream prior to incineration. In Zurich, Switzerland, mercury emissions from incinerators were essentially the same before and after implementation of a public education programme for removal of mercury-containing batteries from the waste stream [8]. Sweden has a similar programme for removal of batteries at source. While the effectiveness of the programme to reduce incinerator emissions of mercury has not been fully evaluated, results from the 2nd and 3rd quarters of 1987 (when the programme started) indicate that approximately 50% of mercury and 20% of cadmium in batteries were collected. The goal is to recover 75% of the mercury in batteries (K. Hasselgren, Swedish Association of Public Cleansing and Solid Waste Management, personal communication, 1988).

Information is limited on the impact on emissions of removing certain waste constituents from burning refuse-derived fuel (RDF). Few plants use waste processing or material separation technologies. In Italy, a very small proportion of plants (approximately 10%) use mechanical processing for sorting metals, plastics, glass, or organic matter (the last processed by composting for agricultural use). Canada has only one mechanical processing system in oper-

ation; it uses shredding and magnetic metal separation. In the USA, many RDF plants have been shut down [9].

Lorber [10] describes a testing programme investigating emissions from combustion of as-received MSW and RDF. Emission tests indicate that the concentration of hydrogen chloride, cadmium, and lead in the untreated flue gas from the RDF combustion unit was in the same range as that found in the flue gas of an MSW incinerator before electrostatic precipitation and scrubbing [11]. These pollutants were removed when the RDF was burned in a cement kiln where the pollutants were fixed in the cement clinker.

In the United Kingdom, experiments are using mechanical sorting in a rotary drum to produce a "cleaner" product to be used in RDF production [12]. Preliminary processing results (Table 3) produced 160 tonnes of product (>40 mm) and 110 tonnes of reject material (<40 mm). Most of the glass, approximately half of the putrescible material, and about one third of the nonferrous metal were removed. A significant portion of the lead and mercury (about two thirds and one half, respectively) from nonmetallic sources was discharged to the rejects (Table 4). Rejection of heavy metals contained within metallic items can only be inferred from the category data (Table 3); a magnetic separation stage would be helpful in this respect. Emissions have not been analysed for heavy metal compounds.

However, considering Lorber's results, one would expect to see a major difference in heavy metal content in the stack emissions. Boiler efficiency was calculated to increase from 68.3% when burning raw waste to 73.2% when burning RDF. To the extent to which this increase in boiler efficiency indicates improved combustion conditions, it may also indicate lower emissions of organics, but no test results are available to evaluate this theory.

Thus, careful consideration should be given to determining, in detail, the trace chemical content of solid wastes before choosing an overall solid waste management strategy.

MSW incineration technology

Waste combustion is a complex process. MSW is heated by contact with hot combustion gases or preheated air and by radiation from the furnace walls. The temperature range for drying is 50 to 150 °C. At higher temperatures, volatile matter is formed by thermal decomposition reactions. This volatile matter is generally combustible and, after ignition, produces flames. The remaining material is further degassed and burns much more slowly.

The combustion of MSW proceeds without supplementary fuel when the heat value is at least 5000 kJ/kg. This occurs when the waste is within the following general compositional limits: ash content, <60%; moisture content, <50%; and combustible matter, >25%.

The complexity of the combustion of solids stems from the nature of the decomposition and burning reactions and their

Table 3. *Analysis of RDF* [12]

Material	Feed	Reject (>40 mm)	Product (<40 mm)
		(% by weight)	
Ferrous metals	6.4	41.0	99.0
Nonferrous metals	0.9	24.0	76.0
Glass	6.4	79.0	21.0
Putrescible material	20.8	30.0	50.0
Textiles	2.6	2.0	98.0
Paper and board	24.3	4.0	96.0
Plastics	7.3	5.0	95.0
Miscellaneous			
Combustible	4.6	5.0	95.0
Noncombustible	4.0	73.0	27.0
Fines (<10 mm)	22.7	92.0	8.0
Total	100.0	41.0	59.0

association with heat transfer, air flow, and diffusion. In most incinerators, combustion takes place while the solids are supported on and conveyed by a grate. Since the early 1960s, most MSW incinerators have incorporated one of a number of proprietary grate systems that allow a continuous feed of unscreened waste into and movement through furnaces with integral boiler facilities. The grate performs several functions: providing support for the burning bed of refuse; admitting underfire air through openings in the grate surface; transporting the solid waste from feed mechanism to ash quench; agitating the bed; and redistributing the burning mass.

Combustion air is introduced as primary air and secondary air. Primary air passes through the grate and the fuel bed and assists in burning the fuel and cooling the grate. Secondary air is blown into the furnace through nozzles above the fuel bed to promote mixing and complete combustion of the burning gases. In some plants, tertiary air is added to control the temperature of the flue gases to required levels before discharge from the combustion area.

During combustion, the heat value of the fuel is converted into high-temperature flue gas and ashes, and into

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Table 4. *RDF product analysis* [12]

Analysis	Feed	Reject (>40 mm)	Product (<40 mm)	% of total	
				Reject	Product
Gross calorific value (MJ/kg)	9.2	6.62	11.0	29.6	70.4
Net calorific value (MJ/kg)	7.91	5.57	9.53	28.9	71.1
Moisture content (% by weight)	31.4	32.1	31.0	41.7	58.3
Ash content (% by weight)	30.5	42.8	21.9	57.6	42.4
Volatiles (% by weight)	31.6	14.4	43.7	18.4	81.6
C (% by weight)	28.8	28.7	29.0	40.8	59.2
H_2 (% by weight)	2.52	1.04	3.55	17.0	83.0
O_2 (% by weight)	12.4	4.9	17.6	16.3	83.7
N (% by weight)	0.71	0.65	0.74	38.0	62.0
S (% by weight)	0.28	0.43	0.18	62.0	38.0
Cl (% by weight)	0.35	0.14	0.49	17.4	82.6
Pb (% by weight)	0.011	0.019	0.006	57.8	32.2
Cd (% by weight)	0.0012	0.0004	0.0017	16.4	83.6
Hg (% by weight)[a]	0.00004	0.00003	0.00004	41.0	59.0

[a]   Identifiable metal pieces were removed from samples prior to chemical analysis. Hence metal content and distribution refer only to nonmetallic sources.

heat losses through the furnace walls. With an estimate of heat losses and the unburned carbon in the residue, the theoretical combustion temperature can be calculated. This temperature may be controlled by introducing varying quantities of excess air into the furnace.

When waste with a low heat value is incinerated or when high combustion temperatures are desirable, heat losses may be kept to a minimum by adequate insulation of the furnace. The combustion temperature may also be increased by preheating the combustion air or by predrying or heating the waste products.

Control of incinerator emissions

Gas cleaning is required to reduce the level of pollutants in

the raw flue gases (Table 5). Electrostatic precipitators (ESP) are the most commonly used gas cleaning device for particulate emission control in municipal incinerators as they can be designed to achieve very high collection efficiencies (99% or higher) and their operating costs are moderate. However, the investment cost and volumetric space required are high.

Fabric filters can operate at high efficiency even in the submicron particle size range. Until recently, they were not widely used in solid waste incineration plants because of their relatively high investment and operating costs, and limited life at high temperatures. However, with the increasing emphasis of regulatory agencies on acid gas control and an apparent trend to lower particulate emission levels, baghouses, usually in conjunction with addition of lime, and lower flue gas temperatures are being used more widely with municipal waste incinerators in the USA.

Wet collectors were initially used to control acid gas. Various semiwet and dry methods of cleaning flue gases have also been used. These methods of gas treatment, based on the injection of slurried or powdered lime, limestone, or dolomite, eliminate a large fraction of the acid gases from the flue gas by adsorption and absorption, followed by chemical conversion. Because the reactivity of these lime materials is rather low, a multiple of the stoichiometric quantity is normally required to obtain a satisfactory cleaning effect. High removal efficiencies can be achieved for hydrogen chloride, but reduction of sulfur dioxide and sulfur trioxide is

Table 5. *Typical composition of raw flue gases (wet basis, NTP*[a]*)*[3]

Substance	Amount
H_2O	10 – 18 % by volume
CO_2	6 – 12 % by volume
O_2	7 – 14 % by volume
CO	< 0.1 % by volume
Dust	2 – 15 g/m^3
Cl^-	250 – 2000 mg/m^3 (as HCl)
F^-	0.5 – 5 mg/m^3 (as HF)
$SO_2 + SO_3$	100 – 1000 mg/m^3 (mainly SO_2)
$NO + NO_2$	100 – 400 mg/m^3 (largely as NO)

[a] NTP = normal temperature and pressure.

only partial. However, slaked lime is highly reactive and stoichiometric ratios of 1.2 to 1.7 have been used for 97-99% of hydrogen chloride removal and 60-90% reduction of sulfur dioxide, depending on operating conditions and particulate collector. Lime injection into a scrubber/fabric filter system has removal efficiencies of 90-99% for hydrogen chloride and 70-90% for sulfur dioxide, provided that the flue gas temperature, certain equipment design features, and stoichiometric ratio for lime addition are suitable.

Test results for MSW incinerators equipped with the conventional two-field ESP have shown a wide range of particulate emissions, varying from 50 to 300 mg/Nm^3 (0.07 to 0.13 grains per dry standard cubic foot (gr/dscf)). The three- and four-field ESP (see Table 6) can achieve emissions of 20 to 75 mg/Nm^3 (0.009 to 0.03 gr/dscf). An emission level below 20 mg/Nm^3 (0.009 gr/dscf) is technically possible and has been demonstrated at Zurich (J.T. Möller, personal communication, 1986) where lime addition was retrofitted on a plant with an existing two-field ESP.

The scrubber/fabric filter control systems have also been shown (see Table 6) to be capable of operating at a particulate emission level of 20 mg/Nm^3 (0.009 gr/dscf) and lower. The material selected for the filter bags can have an important effect on filtering efficiency and the emission level thus achieved. In general, test results for the scrubber/fabric filter indicate lower particulate emissions than those for ESP on MSW incinerators, though ESP have not normally been designed to meet emission levels as low as those specified for fabric filter installations. In Europe, ESP following spray drying absorbers have been tested at particulate emission levels of 1 to 8 mg/Nm^3 (0.00045-0.0036 gr/dscf). The reliability and overall economics of the various control processes must be considered when making a selection of equipment to meet these very low emission control requirements.

Because the condensation point for most metals, such as compounds of lead, cadmium, chromium, and zinc, is above 300 °C, the removal efficiency of such metals depends highly on the removal efficiency of the particulate control system used. Some heavy metal compounds, particularly chlorides, have condensation points below 300 °C. For such compounds, particulate collection temperatures will be a factor in collection efficiency. High removal (over 99%) has been observed for most metals for highly efficient (over 99%) particulate removal systems operating at appropriate temperatures [13].

Traditional incineration facilities have relatively inefficient particulate control equipment and emissions of metals may be relatively high in such facilities (Table 7).

Mercury, a very volatile metal, exists in vapour phase at temperatures as low as 20 to 50 °C. Several studies have suggested that both sufficient cooling of the flue gas (typically below 140 °C, based on test results) and a highly efficient particulate

removal system to remove the particles on which the mercury has been actively absorbed [14,15] (H. Vogg et al., paper presented at Fifth International Recycling Congress, 1986) are required to achieve high mercury removal. High mercury removal has also been obtained using the scrubber/fabric filter and semidry/ESP systems, provided that the flue gas is adequately cooled [13]. Properly operated wet scrubbers are also capable of considerably reducing mercury emissions (Goepfert, Reimer and Partners, unpublished report, 1986).

Data on emissions of PAH and other chlorinated organic compounds are limited. Most sampling programmes for PCDD/PCDF, which have been fairly extensive, have unfortunately neglected to analyse for these compounds. The scrubber/fabric filter technology generally demonstrated removal rates of 80-99% for these compounds [13].

Impact of incinerator operation on atmospheric emissions

An advanced incinerator is designed to achieve 99% conversion of organic matter to carbon dioxide [16]. This objective is achieved by providing adequate residence time, postcombustion temperature, and turbulent mixing. When this objective is achieved, the concentration

Table 6. *Particulate emissions from MSW incinerators* [1]

Plant	Dry filterable material (at 12% CO_2)	
	gr/dscf	mg/Nm^3
G (1983); ESP	0.0321	72
T (1984); DS,FF	0.0120	27
M (1984); DS,EP	0.0104	23
W (1985); DS,FF	0.0040	9
P (1985); ESP	0.0163	37
T (1986); ESP	0.0070	16
M (1986); DS,FF	0.0070	16

[1] Unpublished reports: Cooper Engineers, 1984, 1985; Hahn et al., 1986; Velzy, 1986, 1987).
    DS = Dry or semidry scrubber (lime addition).
    ESP = Electrostatic precipitator.
    FF = Fabric filter (bag house).
    gr/dscf = Grains per dry standard cubic foot.

level of carbon monoxide remains consistently below 0.1% by volume. Generally, levels of 125 mg/Nm^3 (100 ppm) and less are achievable in advanced design plants with modern combustion control systems.

Combustion may be incomplete when the incinerator is improperly operated in the following ways:

- inadequate mixing of the feed material so that large quantities of material with extremely high or low heat value enter the furnace;

Table 7. *Removal efficiencies for certain metals from flue gases from MSW incinerators*

Equipment	Particulate-bound (Cd, Pb)	PAH	Vapour-bound (Hg)
	Removal efficiency (%)		
Traditional			
High eff. ESP[1]	Cd 95–99+		NA
	Pb 95–99+		20%
Advanced			
1. Scrubber + ESP[2] (Semidry)	Cd 98–99++ Pb 98–99++		80–90% (at 120–140 °C)
2. Scrubber + FF[3] (Dry)	Cd 99++ Pb 99++	99 99	85–95% (at 140 °C) Much uncertainty
3. Wet[4]	Cd 99+ Pb 99+		70[a]–90%[b]

ESP = Electrostatic precipitator. FF = Fabric filter (baghouse). NA = Not applicable.
[1] Data from measurements in Europe and North America.
[2] J.T. Möller, personal communication, 1986 and 1987.
[3] R. Bergström, personal communication, 1986.
[4] A. Nottrodt, personal communication.
[a] Dry endproduct 60–75%
[b] Liquid effluent 80–95%

- operation at excessively low temperatures (some regulatory agencies require the use of auxiliary burners when the temperature at the exit from the combustion chamber drops below 800 °C);

- quenching of the flames caused by improper distribution of secondary air;

- gross overloading of the plant.

Incomplete combustion can be detected by monitoring the flue gas composition. Conventional combustion gas measurements include carbon monoxide, total hydrocarbons, and oxygen or carbon dioxide. Both carbon monoxide and total hydrocarbons have been of interest as potential surrogates to indicate incomplete combustion or periods of poor combustion. Under these poor combustion conditions, carbon monoxide levels of several thousand mg/Nm^3 have been observed and total hydrocarbon levels have risen from a typical 1.25 to 6.25 mg/Nm^3 (1-5 ppm) to 125 mg/Nm^3 (100 ppm) and more.

Poor combustion conditions in incinerators can lead to local oxygen-deficient conditions, resulting in the emission of hazardous organic compounds. PAH, for example, are formed during fuel-rich combustion as a consequence of free radical reactions in the high-temperature flame zone. In addition, in the presence of water-cooled surfaces, such as found in oil-fired home-heating furnaces, a high fraction of the polycyclic compounds is oxygenated. Similar free radical reactions probably take place in fuel-rich zones of incinerator flames, yielding PAH, oxygenated compounds (e.g. phenols), and, in the presence of chlorine, some PCDD and PCDF.

The amount of heavy metal compounds, other than those with low vaporization such as mercury, discharged from an MSW incinerator is largely a function of the amount of fly ash discharged. The amount of ash carried with the flue gases from a burning refuse bed increases with increasing underfire air rate and with bed agitation. For this reason, starved or controlled air incinerators with low underfire air flow rates tend to have lower particulate emissions from the furnace than conventional mass-burn units. In addition, the particle size of the inorganic content of the MSW will influence the amount of ash carried from the combustion chamber.

High-efficiency particulate control devices can effectively reduce the atmospheric emission of most heavy metal compounds. As the efficiency of particulate control improves, metal emissions generally decrease and, in most cases, decrease significantly. Even though ample evidence from tests [17,18] (W. Dannecker, personal communication, 1983) indicates that heavy metals tend to concentrate on the finer particles, evidence also suggests that more than 99% metal removal from flue gases can be achieved by using high efficiency (99%) particulate control devices (Cooper Engineers, un-

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published report, 1985; Hahn et al., paper given at National Waste Processing Conference, 1986).

DISCHARGE OF METALS AND PAH FROM INCINERATORS

Organic materials composed entirely of carbon, hydrogen, and oxygen will produce water and carbon dioxide as the final products of complete combustion in the presence of oxygen. However, MSW includes many compounds or elements in addition to the three elements noted above. Therefore, discharges to the environment emanating from the burning of MSW include substances resulting directly from combustion and other substances present in the bed of solid waste. Discharges may be in the form of stack gas and particulate emissions, bottom ash from the furnace grates, and fly ash from the removal of particulates in the flue gases.

Metals are not produced by the incineration process but rather are changed, concentrated or redistributed among the various process effluents. This is in contrast to PAH, which are largely generated as a consequence of incomplete combustion.

Redistribution of elements during combustion

The distribution of elements between bottom ash and ash carried to the air pollution control device depends upon the design and operation of the incinerator and the composition of the feed. The distribution of elements among the different components of refuse also strongly influences the fate of the elements. For example, titanium dioxide, a pigment in paper products, has a particle size of about 0.2 um and will tend to be carried off by the flue gases passing through the refuse bed, whereas titanium dioxide in glass will end up in the bottom ash. About 10% of the inorganic content of the waste will be entrained from burning refuse beds to form fly ash particles [19]. The remainder will end up in the residue.

Volatile elements and their compounds, usually present in trace amounts in the feed, will vaporize from the refuse and condense in the cooler portions of a furnace either as an ultrafine aerosol (< 1 um in diameter) or on the surface of fly ash particles. A large fraction of highly volatile elements in the feed, such as mercury, will be volatilized and emitted to the atmosphere in vapour form unless special preventive measures are taken. Some elements may be converted to more volatile chemical forms during combustion. For instance, the relatively high chlorine content of MSW results in an adequate supply of hydrogen chloride to form substantial quantities of volatile metal chlorides [17] as well as chlorinated organic compounds.

PAH concentrations in MSW incineration residues are relatively uniform (Table 8). However, the metals increase in concentration

from bottom ash, to boiler/economizer ash, to precipitator or fly ash. Greenberg et al. [17] suggest that trace element enrichment on fine particles is primarily due to a volatilization-condensation mechanism similar to that proposed by Natusch et al. [20] for coal-fired power plants. The proportionately greater surface area of fine fly ash particles is expected to result in an enrichment of the volatile trace elements on these particles. However, Wadge et al. [18] suggests that the elevated levels of certain trace elements in incinerator fly ash may result not from a volatilization-condensation mechanism but, in part at least, from the loss of organic fraction in the furnace. This would leave an inorganic residue enriched in trace elements.

Using the information in Tables 7 and 8, the distribution of selected metals and PAH in incinerator residues can be calculated (Table 9). The concentrations of refractory elements such as iron, carbon monoxide and titanium will be higher in the bottom ash.

Atmospheric emissions and deposition

Cadmium, lead, and mercury

A review of data on cadmium, lead, and mercury indicated that emissions were significant (Tables 10-12). The emission levels or toxicities of other metals were so low that they were considered to be of minor importance.

Table 8. *Concentrations of PAH, certain metals, chlorine, and sulfur in MSW-incinerated residues*[1]

Substance	Bottom ash	Boiler/economizer ash	Precipitator ash
PAH (ug/g)	0.1-0.5	0.03-0.2	0.2-0.5
Cd (ug/g)	2.0-7.5[a]	115-350	250-1100[a]
Pb (ug/g)	750-6000[a]	5000-16000	5000-26000[a]
Hg (ug/g)	0.01-1.0[a]	1-8	5-70[a]
Cl (mg/g)	3-6[a]	ND	40-80[a]
S (mg/g)	2-4[a]	ND	20-40[a]

[1] Personal communications: P.H. Brunner, A. Nottrodt, J.T. Möller.

[a] [19,21].

ND = Not detected.

Table 9. Emission factors and distribution (% of input) for selected metals and PAH [22]

Compound	Bottom ash g/t of waste	%	Boiler ash g/t of waste	Fly and boiler ash g/t of waste	%	Fly ash from stack gas cleaning Tradi-tional g/t of waste	Advanced g/t of waste	%
Cadmium	0.4-2	10-40	0.1-4	3-30	55-80	0.4-9	0.1-0.25	1-15
Lead	200-2000	55-80	5-200	50-800	20-45	0.6-300	0.06-5	1-5
Mercury	0.1-0.3	2-6	0.001-0.06	0.05-2	15-35	0.6-10	0.2-1.3	60-85
PAH[a]	20-150	45-12	0.03-2.0	1.5-14	4-1	20-1000		50-80

[a] Distribution in percentage is not based on PAH concentrations in MSW input.

Table 10. *Emission data on cadmium from stacks*

Traditional dry collection 95% +/- efficiency	Advanced acid gas control scrubber (dry or wet/dry) 99% + efficiency
Canada[a]	Canada[a]

0.6–1.0 mg/Nm^3 – 12% CO_2 SA
0.02–1.0 mg/Nm^3 – 12% CO_2
  MB w/ESP
	N.D.–0.6 ug/Nm^3 – 8% O_2
	  MB w/scrubber and FF

France[b]	France[b]

0.06–0.1 mg/Nm^3 – 7% CO_2
  (wet) MB
1–1.5 mg/Nm^3 – 7% CO_2
  (wet) MB
	0.03 mg/Nm^3 – 7% CO_2 (wet)
	  MB w/ESP and scrubber
	0.001–0.003 mg/Nm^3 – 7% O_2
	  (wet) MB, DS and FF

Italy[c]	Italy[c]
0.83 mg/Nm^3 – 10% O_2 (wet) MB	0.02 mg/Nm^3 – 10% O_2 (wet) MB
Switzerland[d]	Switzerland[g]
0.1–2.0 mg/Nm^3 – 7% CO_2	0.01 mg/Nm^3 – 11% O_2
United Kingdom[e]	Federal Republic of Germany[h]

0.16–0.45 mg/Nm^3 – 10% CO_2
  (dry) MB
	00.2 mg/Nm^3 – 11% O_2
	  MB w/SDA and ESP
	0.04 mg/Nm^3 – 12% O_2
	  MB w/wet scrubber

United States[f]	
0.2–0.5 mg/Nm^3 – 12% CO_2 (dry) MB 0.2–1.2 mg/Nm^3 – 7% O_2 (dry) SA 0.5–0.7 mg/Nm^3 – 7% O_2 (dry) RDF w/ESP	

DS = Dry scrubber. ESP = Electrostatic precipitator. FF = Fabric filter. MB = Mass burn. ND = Nondetectable. RDF = Refuse-derived fuel. SA = Starved air. SDA = Spray drier-absorber.
[a] [13,26,27]. [b] Dimitrov, paper given at Third National Seminar of the Air Quality Agency, 1987. [b] [28]. [c] [29]. [d] [30]. [e] [31]. [f] [32]. [g] J.T. Möller, personal communication, 1986. [h] J.T. Möller, personal communication, 1987.

Table 11. *Emission data on lead from stacks*

Traditional dry collection 95% +/- efficiency	Advanced acid gas control scrubber (dry or wet/dry) 99% + efficiency
Canada[a]	Canada[a]
0.6–2.5 mg/Nm^3 – 12% CO_2 MB w/ESP 8.0–15.0 mg/Nm^3 – 12% CO_2 SA	0.002–0.006 mg/Nm^3 – 8% O_2 MB w/scrubber and FF
France[b]	France[b]
34–49 mg/Nm^3 – 7% CO_2 (wet) MB 1.3–1.9 mg/Nm^3 – 7% CO_2 (wet) MB w/ESP	0.055–0.07 mg/Nm^3 – 7% CO_2 (wet) MB, DS and FF 0.78 mg/Nm^3 – 7% CO_2 (wet) MB w/ESP and scrubber
Italy[c]	Italy[c]
19 mg/Nm^3 – 10% O_2 (wet) MB	0.54 mg/Nm^3 – 10% O_2 (wet)
Switzerland[d]	Switzerland[e]
1–18 mg/Nm^3 – 7% CO_2	0.38 mg/Nm^3 – 11% O_2 MB w/SDA and ESP
United Kingdom[f]	Federal Republic of Germany[g]
2.4–10.7 mg/Nm^3 – 10% CO_2 (dry) MB	0.6 mg/Nm^3 – 12% O_2 MB w/wet scrubber; 0.01 mg/Nm^3 – 11% O_2 MB w/SDA and ESP
United States[h]	
0.15–22.0 mg/Nm^3 – 7% O_2 (dry) SA 1.5–1.8 mg/Nm^3 – 7% O_2 (dry) RDF w/ESP	

DS = Dry scrubber. FF = Fabric filter. MB = Mass burn. RDF = Refuse-derived fuel. SA = Starved air. SDA = Spray drier-absorber. [a] [13,26,27]. [b] Dimitrov, paper given at Third National Seminar of the Air Quality Agency, 1987. [b] [28]. [c] [29]. [d] [30]. [e] J.T. Möller, personal communication, 1986. [f] [31]. [g] J.T. Möller, personal communication, 1987. [h] [32].

Table 12. *Emission data on mercury from stacks*

Traditional dry collection 95% +/- efficiency	Advanced acid gas control scrubber (dry or wet/dry) 99% + efficiency
Canada[a]	**Canada**[a]
0.7–0.85 mg/Nm^3 – 12% CO_2 MB w/ESP 0.5–0.7 mg/Nm^3 – 12% CO_2 SA	0.01–0.6 mg/Nm^3 – 8% O_2 MB w/scrubber
Federal Republic of Germany[b]	**Federal Republic of Germany**[b]
0.42–0.62 mg/Nm^3 – 11% O_2 MB w/SDA and ESP	0.07–0.08 mg/Nm^3 – 12% O_2 MB w/wet scrubber
France[c]	**France**[c]
0.5–0.66 mg/Nm^3 – 7% CO_2 (wet)	‹0.06 mg/Nm^3 – 7% CO_2 (wet) MB w/ESP and scrubber ‹0.22 mg/Nm^3 – 7% CO_2 (wet) MB, DS and FF
Italy[d]	**Italy**[d]
0.1 mg/Nm^3 – 10% O_2 (wet) MB	0.01 mg/Nm^3 – 10% O_2 (wet) MB
Switzerland[e]	**Switzerland**[f]
0.3–1.5 mg/Nm^3 – 7% CO_2	0.035 mg/Nm^3 – 11% O_2 MB w/SDA and ESP
United Kingdom[g]	
0.5–0.6 mg/Nm^3 – 10% CO_2 (dry) MB	

Table 12. (cont'd)

USA[h]   USA[h]

1.2-2.2 mg/Nm^3 - 7% O_2 0.028 mg/Nm^3 - 12% CO_2
  (dry) MB w/ESP   (dry) MB w/wet/dry
0.4-0.6 mg/Nm^3 - 7% O_2 scrubber and FF
  (dry) RDF w/ESP
2.0 mg/Nm^3 - 7% O_2 (dry) MB
2.2 mg/Nm^3 - 12% CO_2 (dry) SA

DS = Dry scrubber.   ESP = Electrostatic precipitator.
FF = Fabric filter.   MB = Mass burn.
RDF = Refuse-derived fuel.   SA = Starved air.
SDA = Spray drier-adsorber.
[a] [13,26,27]. [b] J.T. Möller, personal communication, 1987.
[c] Dimitrov, paper given at Third National Seminar of the Air
Quality Agency, 1987. [c] [28]. [d] [29]. [e] [30]. [f] Möller,
personal communication, 1986. [g] [31]. [h] [32].

Cadmium emissions range between 0.06 and 2.0 mg/Nm^3 for traditional plants and 0.001 and 0.04 mg/Nm^3 for advanced plants (Table 10). The lowest specific test result was 0.00004 mg/Nm^3 for an advanced plant [23]. However, as the specific conditions of this plant operation and its design features are unknown, this result should not be included in the range of anticipated test results. Test results for lead ranged between 0.2 and 50 mg/Nm^3 for traditional plants and 0.01 and 0.8 mg/Nm^3 for advanced plants (Table 11). Test results for mercury ranged between 0.1 and 2.2 mg/Nm^3 for traditional plants and 0.03 and 0.2 mg/Nm^3 for advanced plants (Table 12). Results for advanced plants for mercury removal depend largely on temperature and technology.

Table 13 gives ranges of stack emission rates of cadmium, lead, and mercury from incinerators in Europe, Canada, and the USA. To calculate the increase in ambient air concentration given in Table 13, a dilution factor of 5 x 10^{-5} was used. This factor is lower than the one used in the dioxin report of the WHO meeting in Naples [1]. This value corresponds to the value based on the model of the TA-Luft [24]. To calculate the dilution factor for deposition rates, two different values have been chosen:

$$\text{cadmium and lead} \quad \frac{(2 \times 10^{-2} \text{ ug/m}^2/\text{d})}{\text{ug/m}^3}$$

$$\text{mercury and PAH} \quad \frac{(2 \times 10^{-3} \text{ ug/m}^2/\text{d})}{\text{ug/m}^3}$$

The value of 2 x 10^{-2} has been derived from preliminary research on the dry and wet deposition of cadmium and lead in the vicinity of MSW incinerators [25]. The value of 2 x 10^{-3} refers to the model of the German TA-Luft 1986 for industrial incinerators.

In conclusion, the results indicate that traditional MSW incinerators may have a significant impact on the ambient air concentration as well as on the deposition rates of cadmium and lead. In contrast, the contributions from advanced MSW incinerators are in the range of the background levels or even less.

Table 9 summarizes the element emission factors (in grams per tonne of input MSW) and distributions (in percent of input MSW) for the data available for analysis at this meeting. The distribution of the elements depends on the element properties as well as on the design and operation of the incinerator and gas cleaning system, and the composition of the MSW.

PAH

A wide range of PAH has been identified in the various effluents arising from MSW incineration [33,34]. A number of studies have assessed PAH in various parts of incinerators. However, there is a

Table 13. *Emission, ground-level concentration, and deposition rates for selected metals and PAH*

Compound		Stack emissions (mg/Nm^3)	Calculated increase in ambient air concentration (ground level) (ng/Nm^3)[a]	Rural background ambient air (ng/Nm^3)	Calculated increase in deposition rates (wet and dry)[b] $(ug/m^2/d)$	Background deposition rates $(ug/m^2/d)$
Cd	(2)	0.06–2	3–80	<1–5	1–40	0.5–2
	(3)	0.001–0.04	0.05–2		0.02–0.8	
Pb	(2)	0.2–50	8–3000	50–300	3–1000	40
	(3)	0.01–0.8	0.5–40		0.2–20	
Hg	(2)	0.1–2.2	5–10	1–4	0.2–4	1–<1
	(3)	0.03–0.2	2–10		0.02–1	
PAH	(4)	0.003–0.2(5)	0.1–<0.10(5)	<1	0.006–0.4	–

[a] Dilution factor for ambient air concentration, 5×10^{-5} (see text).

(1) Dilution factor for deposition of Cd, Pb (2×10^{-2}), Hg and PAH (2×10^{-3}) (see text).

(2) Traditional incinerator.

(3) Advanced incinerator.

(4) No differentiation between traditional and advanced except as noted.

(5) Large value stands for small-scale incinerator (<500 kg/h), which may produce high air concentration due to low stack height.

lack of consistency in the individual PAH reported and the methods of sampling and analysis used. Consequently, data are insufficient to derive good mass balance determinations that relate the amount of material entering, synthesized in, and leaving the incinerator system.

In a well-run plant, PAH in the refuse feed material are unlikely to persist through the combustion process. More probably, PAH in the various effluent streams result from incomplete combustion of ordinary non-PAH carbonaceous material or high-temperature free radical mechanisms [35,36].

The most important emission pathways of PAH are via the flue gases and the bottom ash, due to the significantly greater mass of bottom ash compared to that of other plant residuals. PAH account for less than 1% of the total organic carbon (TOC) in the products of incineration.

Based on a review of test results, stack emissions were tabulated for PAH and benzo(a)pyrene (BaP) (Table 14). Very little information is available that would allow a separate projection of PAH emissions for "traditional" and "advanced" plants.

The contribution of incinerator emissions to background ambient air concentrations and deposition rates of PAH can be estimated (Table 13) using dilution factors of 5×10^{-5} and 2×10^{-3}, respectively, based on the model of the German TA-Luft [24].

Incinerator ash

During incineration, 98 to 99% of the organic matter of MSW is converted to carbon dioxide and other volatile compounds, resulting in a mass reduction of 65 to 75%. Depending on the composition of the MSW, about 200 to 300 kg of bottom ash is produced per tonne of MSW. The volume of this bottom ash corresponds to about 10% of the initial MSW volume. The amount of filter ash is approximately 10% of the mass of the bottom ash, depending on the operating conditions of the furnace and the gas cleaning system. Ash deposited in the economizer and boiler will vary from 1.0 to 10.0 kg/t of initial MSW. If dry or wet/dry acid gas control scrubber systems are used for emission control, the mass of filter dust or combined fly ash and chemical treatment additive (usually lime) will be increased by 25 to 150%.

As noted earlier, fly ash contains higher concentrations of volatile metals compared with bottom ash (Table 8). Precipitator ash contains higher chloride and sulfur concentrations than bottom ash (Table 8), which together with the smaller particle size and the different chemical form of the element indicates a greater tendency for metals to leach from the precipitator ash than from the bottom ash. Experiments (P.H. Brunner & P. Baccini, paper given at Second International Conference on New Frontiers for Hazardous Waste Management, 1987; S.E. Sawell, unpublished report, 1987) [25] show a

Table 14. *Emission data on PAH from stacks*

Data by country	Technology
Total PAH	
Canada[a]	
0.005–0.02 mg/Nm^3	Traditional MB
0.00002–0.00013 mg/Nm^3	Advanced MB, w/scrubber and FF
0.007–0.012 mg/Nm^3	Traditional MB (starved air)
Italy	
0.0075–0.208 mg/Nm^3	Traditional MB
Japan	
0.04 mg/Nm^3	Traditional MB (batch fed)
0.002 mg/Nm^3	Traditional MB
Sweden[b]	
0.001 mg/Nm^3	Advanced
BaP only	
Canada[b]	
0.008–0.062 ug/Nm^3	One traditional plant
0.000015–0.00013 ug/Nm^3	One advanced plant
Federal Republic of Germany	
0.0005–0.005 ug/Nm^3	Five traditional and advanced plants
Italy	
0.05–33[a] ug/Nm^3	Four traditional and advanced plants

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Table 14. *(cont'd)*

Japan

    0.5–1.5[b] ug/Nm^3 Two traditional plants

United States[c]

    5.9–19 ug/Nm^3 One traditional plant

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[a] BaP and BeP combined data.
[b] Assumed particulate emission rate of 100 mg/m^3.
MB = Mass burn. FF = Fabric filter.
[a] [13,26,27]   [b] [37]   [c] [32]

high leachability at first water contact of the precipitator ash (10% cadmium, 0.5% lead, 20% zinc with deionized water). With a high content of calcium oxide, as used in acid gas treatment, the pH of an aqueous suspension of filter ash rises to 9–10. At this pH, most metals are present as insoluble hydroxides. However, a question remains about the long-term behaviour of ash deposits. In a laboratory study designed to examine long-term leaching patterns at landfill sites, Wadge & Hutton [38] report a relatively rapid movement of cadmium and lead from incinerator fly ash.

    Several factors have the ability to change the mobility of metals in an ash deposit: pH changes, complexing agents, and microbial activity. At pH 3–4, 85% of the cadmium, 5% of the lead, and 50% of the zinc in ash are mobilized (Sawell, unpublished report, 1987) [25]. If lime or other products are added during gas purification, the pH and the salt content change, which alters the physicochemical behaviour of the filter residue (mobility of the amphoteric elements). Thus, to ensure safe long-term behaviour in a natural environment, further treatment of flue gas cleaning residues may be required.

    The content of organic matter in the filter residues is about 10 to 40 g/kg. Less than 1% of the compounds in this fraction has been identified. Of those identified, the largest group consists of chlorinated benzenes (approximately 1 ug/g), PAH (0.2–0.5 ug/g), and chlorinated phenols (approximately 0.5 ug/g) [16]. The organic compounds are tightly bound to particle surfaces and are not mobile in an aqueous environment if organic solvents are absent. An exception is chlorinated phenols, which may go into solution. Still unknown is the extent to which carbon can be used by microbial activities, reducing pH levels and changing the leaching characteristics of ash

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deposits (P.H. Brunner & P. Baccini, paper given at Second International Conference on New Frontiers for Hazardous Waste Management, 1987).

Research on disposal of MSW incinerator ash residues addresses the following topics:

- concentration of the hazardous elements in the ash, including controlled leaching of the metals at different pH;

- fixation of the metals in the residues by vitrification or solidification (cementor silica-based) [39,40];

- destruction of carbonaceous residues by optimizing combustion and reheating filter dust;

- leach tests to determine the long-term behaviour of residues from MSW incineration in the natural environment;

- plant availability of metals in incinerator residues [41].

Government regulations

Canada, the USA, and most countries in Europe have regulations covering emissions from MSW incinerators. Regulations normally establish specific control levels for the emissions of, for example, dust, carbon monoxide, oxides of nitrogen, heavy metal compounds, and acid gases, together with stipulations for the method of monitoring and enforcement.

Pollution control agencies consider a number of factors that influence their decisions on how to control MSW incineration, including the following ones:

Plant-related factors. Some countries, such as France, feel that large plants should be more strictly controlled than small ones. Similarly, plants within residential areas may have to be regulated more carefully than those well removed from towns. Other factors within this category would include the type of plant, local topography, and prevailing meteorological conditions.

Pollutant-related factors. Pollution control decisions will also depend upon the nature of the substances to be controlled. Some, such as specific metal compounds, can be relatively easily sampled and analysed directly. Others, such as PAH, are very costly to determine as individual compounds within this entire group of compounds. Even with metals, it is arguable whether those metals of similar toxicity should be identified and controlled as a group, as is the case in France, rather than establishing emission limitations for individual metals as is generally the case in the USA. Also,

while the total amount of a metal is generally determined in a test, sometimes the species are of interest.

Pollution control options. Another choice to be made is whether action should be taken to control the process using specific control limits or by specifying the equipment technology to be used.

Control limits can be applied either at the source directly or as ambient air quality standards at the point of maximum source impingement on a receptor. When applied at the source, operators are allowed to choose the technology to control their plant emissions, provided that set control limits are met. These limits would specify the maximum quantities of a substance that could be emitted by any emission route, whether it be to air, water, or land. In the case of a metal, the control limit could relate directly to the pollutant, as is the case in many countries. In the case of PAH, which are difficult to measure as a total listing of all compounds, indirect indicators might have to be used as surrogates. These are analytes (such as BaP), which are relatively easy to determine and vary in concentration as the target species would. This approach has some attractions, but the identification of suitable surrogates is difficult and, in the case of PAH, would require further research work.

In some cases, ambient air quality standards have been adopted: for example, the Council of the European Communities Directive on the limit value for lead in the air (OJ L 378, 31 December 1982). This approach is useful when the pollutant is emitted from a large number of small sources.

Specific control procedures are a commonly used alternative or addition to setting control limits. The control equipment or technology that an operator should adopt is specified (i.e. best available control technology, BACT), bearing in mind that this should not entail excessive costs. This approach is being adopted increasingly by the Council of the European Communities. It is also somewhat similar to the best practicable means (BPM) system used in the United Kingdom.

When both the substances to be controlled and a well-defined control technology can be identified, legally binding emission standards can be set, as many countries have with metals. But in the case of PAH, the compounds most hazardous to health can probably not be identified or a clear consensus arrived at as to the most cost-effective control process. In these circumstances, recommendations or guidelines are commonly used. While they are not legally binding, they do indicate target values to be aimed at. This approach has proved useful for dioxin control in the past (i.e. Sweden, Denmark, and Italy) and has encouraged interested parties to identify the control issues and adopt appropriate control practices.

An implicit component of any control strategy is the need to test whether a control requirement is being met; this is called compliance testing. Broadly speaking, compliance testing has two main approaches: monitoring to check emission levels and/or visits by qualified inspection personnel.

The problems of monitoring emissions are considerable and should not be underestimated. Neither should monitoring instrumentation and procedures be assumed to be sufficiently developed to be accepted (as proof of plant malfunction) in a court of law.

Both regulatory pressures and technological advances have succeeded in transferring the environmentally harmful species from stack emissions into the water effluents and residual ashes from MSW incinerator. The legislation for the disposal of these effluents is often less well developed than those for air pollution control and may have to be modified in the future.

Regulations for MSW incinerator emissions

Many countries have national regulations for controlling emissions of heavy metal compounds. These are constantly under review, and listing the provisions of all the countries is impossible. Instead, the current proposals of the Commission of the European Communities for new municipal waste incineration plants can be given as an indication of possible regulations (Table 15). The emission limit values in Table 15 are standardized at the following conditions: temperature, 273 K; pressure, 101.3 KPa; and 11% oxygen or 9% carbon dioxide (17% O_2 in case of incineration plants of a nominal capacity less than 1 tonne per hour).

Water and solid effluent provisions

Incinerators produce liquid effluents from ash quench tanks, wet scrubbers, and other areas of water usage in the processing of MSW. They also produce large quantities of residual ash for disposal. Many countries have guidelines for the handling and disposal of these effluents, but few have specific legislation that treats solid and liquid effluents from MSW incinerators differently from any other pollutant discharge. While a general discussion of water treatment and solid waste disposal legislation is outside the scope of this report, the specific problems of MSW ash handling have prompted some governments, including France, the Federal Republic of Germany, Sweden, Switzerland, and the USA, to consider legislation calling for specific regulations for MSW incineration plants. These proposals range from requirements to contain and treat wastewater produced by MSW incineration prior to discharge, to requirements for the segregation of ashes and the separate storage/disposal of fly ash in special landfills.

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Table 15. *Proposals of the Commission of the European Communities*
for regulations for MSW incinerator emissions

Pollutant	Emission limit value[1] (mg/Nm^3)
Total dust	50[a]
Heavy metals	
Pb + Cr + Cu +Mn	5[b]
Ni + As	1[b]
Cd + Hg	0.2[b]
Hydrochloric acid	50[a]
Hydrofluoric acid	2[b]
Sulfur dioxide	300[b]

[1] These values do not apply to plants designed to burn refuse-derived fuel and that use such fuels exclusively. The Commission shall submit to the Council as soon as possible proposals for limit values to be applied to these plants.

[a] 100 mg/Nm^3 in case of incineration plants of a nominal capacity less than 3 tonnes per hour.

[b] These limits do not apply to incineration plants of a nominal capacity less than 1 tonne per hour.

HEALTH-RELATED BACKGROUND

Major episodes of ill-health have been reported among populations acutely and chronically exposed to cadmium, lead, and mercury arising from non-occupational sources other than MSW incinerators [42-44]. Epidemiological studies have not been conducted in the vicinity of MSW incinerators, but considerable amounts of cadmium and mercury are emitted from these facilities [30,45,46] and lead is a major constituent of the metals in MSW [47]. For these reasons, the metals of prime concern with respect to emissions from MSW incinerators are cadmium, lead, and mercury.

This section summarizes the environmental pathways, metabolism, toxicity, and human health effects of these three metals. Comprehensive reviews of their health effects are available [42,48-51]. The relative significance of metal and PAH emissions from MSW incinerators compared with other high-temperature combustion and industrial processes is considered later.

Human health concerns of cadmium, mercury, and lead

Exposure pathways

Generally, dietary intake is the major route of exposure for the three metals. In the case of lead, inhalation is also an important source of exposure in urban-dwelling populations [51]. Infants may also ingest considerable quantities of lead by inadvertently consuming dusts or deliberately ingesting lead-rich paint or soil [52,53]. Tobacco smoking can be an important source of cadmium intake [54]. Methylmercury is the chemical species of mercury of importance in environmental exposure, and, in the general population, the major ($>$90%) source of this is fish consumption [55].

The three metals differ greatly in the percentage that is absorbed across the gastrointestinal tract. Methylmercury is absorbed efficiently (approximately 95%) while the fractional absorption of cadmium is only about 5%. This latter value increases considerably in cases of iron deficiencies [56]. Lead absorption is often reported to be about 10% in adults, but a higher value (approximately 50%) is often assumed in children [57,58].

Metabolism

Once absorbed, the three metals display characteristic tissue distributions. Cadmium selectively accumulates in the kidney and liver where it binds to the protein metallothionein [42]. In contrast, methylmercury is initially distributed throughout the body, readily crossing the blood-brain barrier, although the kidneys retain the highest tissue concentration [59]. Most of the lead (approximately 90%) is stored in the skeleton, but the liver and kidney also contain relatively high concentrations. An important feature of lead and methylmercury is that both easily cross the placental barrier; this is particularly efficient for methylmercury. The placental barrier is more efficient for cadmium; the median blood cadmium of neonates was about 50% lower than in mothers [60].

Health effects and critical populations

Lead may affect two endpoints: the haematological system and the central nervous system. The haematological effects arise at the beginning and final steps of haem synthesis. Interference in the last stage results in the accumulation of protoporphyrin in the blood, which may occur at blood lead concentrations of about 15 ug/100 ml; that is, at concentrations regarded as "normal". At higher levels of lead exposure, anaemia can develop. The haematological effects of lead are reversible.

Of potentially greater concern is the reported association between certain neurobehavioural effects and lead exposure in ur-

ban-dwelling children [61]. Decreases in measurements of intelligence and behaviour have been found in children from the general population subjected to commonly occurring levels of lead exposure. These effects are widely believed to reflect the greater susceptibility of the fetus and child to lead resulting from greater sensitivity of the developing central nervous system and higher intake and uptake of the metal as compared with adults.

Methylmercury intoxication is also characterized by effects on the central nervous system, the major areas affected being associated with the sensory, visual, and auditory functions as well as those concerned with coordination [55]. The developing nervous system of the fetus is more sensitive than that of the adult, and prenatal exposure can subsequently result in neurotoxic effects in the infant in the absence of effects in the mother. Populations with heavy fish consumption have been identified as being exposed to elevated levels of methylmercury, which is enriched to a high degree in the aquatic food chain [55].

The kidney is the critical organ of intoxication after long-term exposure to cadmium [42]. Increased urinary levels of B_2-microglobulin (B_2M), retinol-binding protein (RBP), and N-acetyl-B-D-glucosamidase (NAG) appear to be the most sensitive parameters of early changes in tubular function. The concentrations of B_2M in serum and albumin in urine appear the most sensitive parameters of changes in glomerular function. However, these forms of proteinuria are not specific to cadmium [62-64]. Nevertheless, measurement of B_2M is being increasingly adopted to monitor for the early effects of cadmium exposure.

Human health concerns of PAH

Environmental data and exposure pathways

About 500 PAH have been detected in air (Pott, address before the Federal Council of Environment, Berlin (West), 1983), but reports of ambient air levels of PAH dating from the 1950s to about 1970 are limited only to BaP only. These data are not necessarily comparable with more recent reports because many different sampling procedures and analytical methods have been used. The natural background level of BaP is determined by the production of PAH in vegetation [65]. PAH could be detected even in borings out of 140-m depth more than 100 000 years old [66]. Compared with the anthropogenic load, the natural background level of PAH in air amounts to not more than 1%.

The annual average of BaP concentrations in urban European locations often exceeded 100 ng/m^3 in the 1960s [67], while more recent measurements in the Federal Republic of Germany and USA indicate that levels are now mostly below 10 ng/m^3 [67,68].

Daily intake of BaP by inhalation of polluted ambient air containing a relatively high concentration of 50 ng/m^3, which is similar to the levels found in the vicinity of coke oven plants [69], is estimated at 400 ng BaP [51]. However, the BaP intake in clean rural areas and cities where adequate smoke control has been achieved may be only a few percent of this or less. Tobacco smoke will add to the inhalation of PAH. Modern "low-tar" cigarettes deliver 10 ng BaP [70], while Grimmer et al. [71] measured 22 ng/m^3 of BaP in a room heavily polluted with cigarette smoke.

Concentrations of BaP in drinking water range from 0.1 to 23 ng/l [72]. The possible sources of PAH in foodstuffs are varied and extremely widespread. Curing smokes, contaminated soils, polluted air and water, modes of cooking, food additives, food processing, and endogenous sources have been investigated. Some 100 PAH have been identified and numerous reviews are available [72-74]. BaP concentrations up to 50 ug/kg were measured in broiled and smoked food [73], and BaP concentrations ranging from 2 to 540 ug/kg have been reported in shellfish [75]. A person in the German Democratic Republic is estimated to ingest 24 to 85 mg BaP during his or her lifetime (70 years) [72]. The daily intake of BaP from food is estimated to range from 0.16 to 3.0 ug in the USA [76].

For most people, oral intake of PAH is considerably greater than the amount inhaled. Much attention has been given to the role played by inhalation of PAH in causing respiratory tumours, but the evidence is not clear whether, and to what extent, ingested PAH may be implicated in respiratory or other tumours.

Uptake and metabolism

PAH are highly lipid-soluble and are absorbed across the lungs, gut, and skin of mammals. As most of the PAH in the ambient air are attached to fine particles, the potential health effects of PAH after inhalation depends on the deposition rate in the respiratory tract, residence time in the lung, the dissolution rate of PAH from the particles, and the chemical structure. Furthermore, the overall composition of the particulate matter (i.e. the ratio of the mass of the carbonaceous core to the mass of the attached organic material) is a factor in determining the biological half-life of the particle-related PAH in the lungs. Particles that consist mainly of organic material, such as coal tar aerosols, will be eliminated from the lungs almost completely within 24 hours. In contrast, the lung clearance half-life of soot particles, which consist mainly of carbonaceous material, such as diesel soot, is relatively long [77]. PAH associated with the latter particle types may exert a stronger carcinogenic effect in the lungs because their residence time is longer.

In general, PAH metabolism consists of a series of oxidation steps catalysed by monoxygenases and epoxide hydrase. Metabolites

include epoxide intermediates, dihydrodiols, phenols, quinones, and their combination. After being metabolized to dihydrodiols, some of the epoxides can be further oxidized to diol epoxides. These latter compounds are generally thought to be the ultimate carcinogens [72].

Oxidized metabolites of PAH are later conjugated with glucuronic acid, glutathione and sulfonic acid, forming polar metabolites that are more easily excreted from the body [78,79].

Toxicity data and biological effects in laboratory animals

Toxic effects of PAH have not been studied as well as their oncogenic effects. Longer-term oral exposures to PAH have resulted in various systemic effects in animals, but overt signs of toxicity are not usually produced until the dose is sufficiently high to produce tumours [72].

Although embryotoxic, teratogenic, and immunosuppressive effects have been observed in some assays, PAH are not acutely toxic compounds.

Several PAH and some of their metabolites have been shown to be mutagenic in mammalian cells and bacterial assays [72,78-80].

The ability of individual PAH to produce neoplasia has been intensively researched for some 50 years, particularly on induction of skin tumours in rodents by topical application of PAH. The carcinogenicity of selected PAH in rodents by different routes of administration has been recently summarized [81].

BaP-induced tumours usually develop at the site of application. For example, rats fed BaP develop forestomach tumours [82] while intraperitoneal infection leads to hepatic tumours when phenobarbital, a promoter, is added to the diet [83], and inhalation induces respiratory tract tumours [84]. Respiratory tract carcinogenicity of BaP is enhanced when this compound is coadministered with particulates, such as ferric oxide, or is absorbed to the administered particles [85].

Many substituted PAH, such as nitro-, methyl- and oxygenated PAH, are also likely in combustion products. Many of these compounds are mutagenic in in vitro tests, and some have been reviewed for evidence of human carcinogenicity [72].

Human health effects

Data summarized by IARC [49] demonstrate carcinogenicity to humans from industrial exposures to mixtures containing PAH. These occupational exposures include aluminium production, coal gasification, coke production, and iron and steel founding. Bitumens, coal tars and derived products, shale oils, and soots have also been evaluated for their human health effects (skin cancer) [86]. A detailed evaluation of the carcinogenicity of PAH in coke oven emissions has been published [87].

Epidemiological studies of these workplaces provide sufficient evidence of the role of inhaled PAH in inducing lung cancer [88-92]. However, levels of PAH in these workplaces were very high, often exceeding 10 ug/m^3 BaP [49,93,94], which is several orders of magnitude greater than in ambient air. In addition to high PAH levels, other carcinogenic gases and metals were present in the occupational settings studied. For example, coke oven emissions may contain other carcinogenic agents, including arsenic, chromium, nickel(VI), 2-naphthylamine, and benzene.

Cigarette smoking is strongly and consistently associated with cancer at several sites, including the lung, larynx, oral cavity, oesophagus, bladder, and kidney [95,96]. Thirty-four PAH, some of which are recognized carcinogens, have been identified in mainstream and sidestream cigarette smoke [72].

Little is known about the possible implications for human health of PAH in food and water, and the role of PAH in toxic endpoints other than carcinogenicity has not been well studied. Therefore, on the basis of available experimental results, the carcinogenic potential of PAH is considered the most significant health effect for the general population.

Characterization of potential human health risk from PAH

A major difficulty in hazard assessment of PAH is that the data for compounds other than BaP are very limited, and that from most emissions, such as those from municipal waste combustion, the full range of PAH has not been analysed. The hazard assessment of this diverse but related group of components would benefit from a model of biological effects compared with those of a well-studied compound, as has been done for dioxins. To this end, several attempts have been made to compare carcinogenic activity of various PAH with that of BaP [81]. At present, however, such a comparative ranking seems prudent only for the most general of classification.

Because several PAH are carcinogenic [49,87] and data for BaP are most extensive, BaP is normally used as an index of the carcinogenic potential of all PAH in a sample. However, PAH profiles detected in different types of industrial emission and workplace air can differ widely.

Several approaches have been taken to provide quantitative estimates of potential human risk from PAH exposure. WHO [51] calculated a unit risk of 9×10^{-5} cases of cancer for a lifetime exposure to 1 ng/m^3 BaP. This upper bound of estimates is derived by application of the linearized multistage procedure to data for lung tumour incidence among coke oven workers [81,91] and using a reported value of 0.71% BaP in benzene-soluble coke oven emissions [97]. This risk assessment implies that about 9 per 100 000 exposed people may die from cancer of the respiratory tract as a result of spending a lifetime in ambient air containing an average level of

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1 ng/m^3 BaP in a mixture with all the other PAH and related substances of coke oven emissions.

SIGNIFICANCE OF MSW INCINERATOR EMISSIONS FOR HUMAN HEALTH

Emissions from MSW incinerators, in common with many high-temperature combustion and other industrial processes, contain a wide range of inorganic and organic compounds, including metals and PAH [31,32,34,45]. When assessing potential health hazards, emissions of metals and PAH from incinerator plants must be considered in conjunction with pollution from other pertinent sources in the area. In Czechoslovakia, for example, coal combustion has been identified as an important source of this type of emissions [98-100].

This section considers the significance of MSW incinerator emissions for human health Occupational hazards are not considered here, but workers in incinerator plants will be exposed to dust and fumes containing metals and PAH [101]. The problems encountered will be similar to those in other industrial plants where furnaces are operated.

Technological aspects

MSW incinerators initially included less efficient air pollution control equipment than required by most current regulations. Thus, the emissions of pollutants were high, as can be seen from examples cited in Tables 10-12 and 14. The impact of such emissions on the environment can be assessed only in rough terms, using data on (a) the likely concentrations of toxic chemicals in the vicinity of the plants, resulting from modelling computations, (b) actual measurements of the concentrations in the air close to incinerators as compared with the background levels, and (c) measurements of toxic chemicals in soil and vegetation near incinerators.

In cases (b) and (c), measurements at various distances from the plant are helpful in evaluating the real impact, that is, the increase above the local background level.

Although such data on the impact of traditional incinerators are incomplete, they point to the possibility of some impact on the environment in the immediate vicinity of the plant. Such assessment, even if incomplete and based largely on modelling computations, justify legislation restricting emission from MSW incinerators. Advanced control technology can markedly reduce the emission of metals (Table 7). The following considerations seem to be relevant in this respect:

   (a) the air concentrations (increase above background level) in the immediate vicinity as compared with the urban back-

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ground levels; here, the direct toxic action via inhalation should be considered

(b) the indirect impact in the vicinity of the plant, the indicators of which may be the increase of the concentration in soil and plants grown for human consumption

(c) direct impact in the immediate vicinity on humans, as judged by biological monitoring of human samples (e.g. blood, urine, and hair)

(d) indirect impact at regional and global level, assessed by comparing of emission sources of varying strength.

Populations under consideration

Two distinct populations can be identified when considering exposure to refuse incinerator emissions. The first consists of those individuals who are close to incinerator plants while the second is the general population.

Populations close to incinerators

In many countries, refuse incinerators are located in densely populated areas. Therefore, the identification of the potential routes and intensity of exposure of local residents as a result of emissions from these plants are pertinent. Unfortunately, little information is available on the environmental levels of cadmium, lead, mercury, and PAH around municipal incinerators. However, their emissions could elevate air concentrations, deposition rates, and soil and dust levels. The deposition of mercury is especially difficult to predict because of the mobility of this element in the environment through re-vaporization that gives rise to the "bounce off" effect reported for other point sources [102]. Mercury emissions may increase mercury levels in local freshwater, especially in water systems with low pH. Elevation of ambient air levels of metals is unlikely to be large near advanced incinerator plants for several reasons: stringent emission controls are imposed on new incinerator plants; modelling data indicate minimal impact around advanced incinerators; and high urban background values may mask any contribution from refuse incinerators.

Air concentration increases will have to be relatively large to increase noticeably total exposure levels, as the dietary intake of these metals is normally greater than the respiratory intake. This is particularly important for cadmium where even a doubling of urban background levels (approximately 10 ng/m^3) would produce only minor increases in total cadmium exposure.

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Deposition of cadmium, lead, and PAH onto soil and garden vegetables around incinerators, either from stack emissions or fugitive release, could increase dietary exposure. Once again, little information is available.

Finally, street and possibly household dusts could contain elevated concentrations of metals (cadmium and lead) and PAH as a result of incinerator emissions. Lead levels in urban dust are markedly elevated (approximately 1000 ug/g) compared with those of rural dust, and incinerator emissions are unlikely to cause detectable increases, although fugitive releases may have an impact in the immediate vicinity. The relative contribution of incinerators will increase only when the use of unleaded petrol is widespread.

General population

The exposure pathway of relevance with respect to the general population is dietary uptake. The tall stacks (commonly about 100 m) of modern incinerators will widely disperse and dilute the plume. Thus, the metal and PAH emissions may contribute to the deposition of these contaminants at considerable distances from the source. The deposition of contaminants on agricultural land or in water systems can have two effects: a long-term increase in the soil or sediment concentration of the contaminant in question; and a direct deposition onto crop plants and the subsequent ingestion of the contaminant. In the first case, if the soil level increases, the crop level may also rise. This is the case for cadmium, but soil uptake of lead and PAH by plants is considered to be less important than direct deposition [103].

The question needs to be answered, however, whether incineration contributes significantly to the deposition of these contaminants at sites distant from point sources. Such an assessment requires information on the emission strengths of other sources to construct an inventory of atmospheric release. In the case of lead, motor vehicle emissions dwarf incinerator emissions [104]. Therefore, this pathway is unlikely to be of current importance in Europe for lead in crops. The situation for PAH is uncertain given the lack of information. Only for cadmium is the pathway considered important because of the relatively large discharges of this metal from incineration [105].

Local health impact of cadmium

The Air Quality Guidelines for Europe [51] quote a concentration of 2 ug/m^3 of cadmium (2000 ng/m^3) as being likely to cause, through life-long exposure, renal accumulation of this metal sufficient to reach the critical concentration in the kidney. The recommended levels were are <1-5 ng/m^3 for rural areas and 10-20 ng/m^3 for urban areas, with emphasis on the qualifying

statement that "an increase in present levels should not be tolerated".

Direct measurements of cadmium concentrations in the air close to incinerators seem to be lacking. A rough estimate can be obtained in two ways. One way is through modelling, in which one assumes for the ground level a fraction of the cadmium concentrations directly measured in the effluent gases. Such calculations, although burdened with a high uncertainty, lead to the expected increase (over background levels) of the order of up to 80 ng/m^3 for traditional incinerators and less than 2 ng/m^3 for advanced incinerators (Table 13). Such values, neither alone nor in combination with the background levels for urban areas, would be expected to lead to adverse health effects by inhalation. The second way is through recomputing the existing data on the rate of deposition. In a pathway analysis, 1 ng/m^3 is estimated to cause a deposition of 1.6 g/ha/y [106].

Deposition rates of cadmium in the vicinity of an incinerator [25] in a rural area of the Federal Republic of Germany were about 20 g/ha/y for a traditional incinerator, and half this value after advanced control equipment was added. The first value corresponds roughly to an ambient air concentration of 10 ng/m^3. Further similar examples could be considered, but the conclusion would be valid that these concentrations are at least one order of magnitude lower than the critical dose for inhalation. The above example, if applied to an urban area where the background concentrations of cadmium are generally about 5 to 10 ng/m^3, would lead to a concentration between 10 and 20 ng/m^3, and the conclusion remains valid.

The possible influence of incineration on environmental levels of cadmium in the immediate vicinity of the plant has, until recently, received little attention. Hutton et al. [107] record, in the vicinity of a traditional incinerator in the London area, deposition rates of about 3 to 70 g/ha/y in the downwind area as compared with 1 to 16 g/ha/y in the upwind area. This deposition has not, after 15 years of operation, markedly elevated cadmium in either street dust, soil, or grass as compared with the upwind area and other London locations. Studies summarized by Brunner & Zobrist [30] reported elevated levels of cadmium in spinach grown near two refuse incinerators, though only trace quantities were detected in grass samples near other plants.

At present, the evidence does not indicate that the level of contamination of the environment by cadmium in the immediate vicinity of incinerators has created a health hazard. However, only limited data are available and this situation may change with time. Therefore, locally grown food should be periodically monitored for cadmium levels.

Local health impact of mercury

Urban ambient air concentrations of mercury are usually in the range of 4 to 5 ng/m^3 [55]. For occupational exposure, a concentration in air of 50 ug/m^3 is allowed. For non-occupational exposure, the outdoor and indoor air concentrations of mercury are proposed to be safe (with an appreciable safety factor) at 1 ug/m^3 [51].

The concentrations of mercury in ambient air in the vicinity of incineration plants have not been monitored. However, concentrations are unlikely to exceed 100 ng/m^3 (see Table 13) and cannot be considered of direct importance to the health of the inhabitants, irrespective of age, sex, and health status.

Inclusion of mercury in this report is based solely on one pathway. This involves deposition of mercury from the atmosphere onto soil and into surface waters: only the latter is relevant. Here, through conversion into methylmercury, it accumulates in fish that are eaten by people.

The only situation in which this pathway could have local impact is an area with rivers or lakes in the vicinity where fishing is an important activity and where the fish are consumed locally. The exposure assessment will be constrained by several factors, such as the history of past contamination of the area from other sources (as mercury in the lake sediments has a long residence time) and the background of the given area (region) dependent on the global mercury cycle, which basically dominates the deposition of the metal. Another confounding factor, however, is the acidification of surface waters increasing in parallel with the overall contamination of ambient air from anthropogenic sources, enhancing the solubility of mercury deposits [102].

Local health impact of lead

Inhalation represents an important source of lead uptake for the general adult population. Uptake by inhalation is in the range of 20 to 70% of the total, depending on the ambient air concentration. Based on the assumed total uptake, the Air Quality Guideline value for lead in ambient air is 0.5 to 1.0 ug/m^3 and this includes a safety factor of about 2 [51].

The ambient air levels of lead in European areas vary from 0.1 to 0.3 ug/m^3 in rural areas and from 0.5 to 3.0 ug/m^3 in most European cities [51]. The degree to which waste incineration increases these concentrations is not known, and some estimates could be made only from modelling data and the few existing data on lead deposition rates.

In rural areas of the Federal Republic of Germany, the background deposition rate of lead is 150 g/ha/y, whereas downwind at <1 km of an incinerator, this value is 500 g/ha/y for the traditional

type and about 275 g/ha/y for the same plant after modernization [25]. Assuming that these data on lead concentrations in ambient air in rural areas are of the order of 0.1 ug/m^3, the lead levels near an incinerator (located in a rural area) could be of the order of 0.2 to 0.3 ug/m^3. Such values, if applied to an urban area, would elevate the background levels by some 0.1 to 0.2 ug/m^3. Thus, waste incineration alone does not seem likely to cause locally significant elevations of the ambient air lead concentration sufficient to lead to health effects by inhalation.

Other available data [107] do not point to significantly elevated levels of lead in street dust, soil, or grass in the vicinity of an incinerator in the London area. No impact on human health from the deposition of lead from incinerators onto locally grown food is expected.

For children, general contamination of the environment (e.g. street dust and soil) may play a dominant role in their exposure pattern. At present, the increased exposure derived from incinerator emissions via this pathway cannot be quantified but it is not expected to play a major role in urban areas.

To summarize, in general no local health impact of lead from incinerator emissions alone should be envisaged. The relative significance of lead emissions from incinerators to total exposure levels among local populations will be greater in more rural areas, where other sources, particularly motor vehicle emissions, make a smaller contribution.

Contribution of incinerator emissions to metal exposure in the general population

This section assesses the extent to which MSW incineration influences metal exposure in the general population. The issues of concern are the fate and significance of the long-range transport of emitted metals and the metal build-up in soils and sediments in rural areas.

Mercury

Fish consumption is by far the major source of methylmercury exposure in the general population. In most countries, marine fish are the dominant source but in some countries, freshwater fish are also regularly consumed. MSW incineration is a significant source of inorganic mercury in the atmosphere, but in most industrialized countries, coal combustion and other sources (e.g. chlor-alkali works) dominate. Nevertheless, MSW incineration is expected to make an important contribution to the mercury cycle at the regional and global level. What is not clear, however, is the significance of this perturbation on the behaviour and accumulation of methylmercury in edible fish species. Direct discharges of inorganic mercury to

aquatic systems are expected to be more important in this respect and to conceal any effect from atmospheric inputs, at least in the short term. Continued release of large quantities of airborne mercury by human activities may progressively increase concentrations of mercury in freshwater and possibly marine systems. If this is the case, MSW incineration is only one contributor to the overall anthropogenic total of airborne mercury; coal combustion is much more important.

Cadmium and lead

The issue of concern is whether airborne emissions of cadmium and lead from MSW incineration bring about significant atmospheric inputs of these two metals to agricultural land. For both metals, dietary intake is a major source of exposure for the general population. Even in rural areas, direct atmospheric deposition is an important source of lead in certain crop species [108].

However, most of the lead deposited in such areas is derived from petrol combustion [104]. Thus, refuse incineration will make only a minor contribution to lead levels in crops in rural areas.

For cadmium, the situation is different because, in contrast to lead, plants take up cadmium relatively efficiently from the soil [103]. Thus, the accumulation of cadmium in agricultural soils is of particular concern as this can lead to enhanced crop concentrations and hence increases in dietary intake for the general population. Important sources of cadmium inputs to arable land are phosphate fertilizers and atmospheric deposition. At local level, sewage sludge application is also important. In the United Kingdom, one third of the contribution from atmospheric deposition is estimated to originate from MSW incineration [103].

Many countries have taken steps to reduce cadmium emissions, while others are trying to prevent any increases in the average dietary exposure to the metal. At the same time, countries are increasingly recognizing that, in contrast to lead, the multiple source nature of cadmium necessitates control measures on a wide range of sources to reduce effectively inputs to agricultural land.

In view of this, refuse incineration is considered by several authorities to be a sufficiently large source of cadmium to merit further controls.

Environmental release of metals from refuse incineration compared with other sources

Numerous human activities result in the environmental release of potentially toxic trace elements. The identity of these sources is well known, and attempts have been made in recent years to assess their quantitative importance. Such studies have been carried out at the local level for a single city or county as well as at the

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regional and global level. The inventories produced by these in-
vestigations provide an insight into the relative importance of
traditional refuse incineration as an environmental source of metal
release. These studies refer to emissions from traditional MSW in-
cinerators; the contribution from incineration to overall metal
emissions will decline as incinerators of advanced design become
widely used.

Environmental release of metals in the United Kingdom

About 34×10^6 t of MSW are generated annually in the United
Kingdom, of which about 10% is incinerated. Emission factors (gram
metal released per tonne material consumed) were derived from ex-
perimental in-stack studies of incinerators in Europe and the USA.
Table 16 shows the resulting atmospheric emission discharges. The
residual trace metal burdens associated with the fly ash and furnace
clinker are landfilled. Table 16 also shows the estimated quantities
of the trace metals present in the remaining 90% of the MSW that is
landfilled directly.

The data in Table 16 may be compared with the inventories of
trace metal release to the atmosphere and landfill for all anthropo-
genic sources in the United Kingdom. In the case of cadmium,
Table 17 indicates that traditional refuse incineration is the
single largest source of atmospheric release, being responsible for
about one third of the total. Atmospheric lead discharges are also
considerable from traditional incineration, being greater than
emissions from nonferrous metal production. Nevertheless, lead
emissions from petrol combustion currently dwarf all other sources
of lead. Mercury emissions from incineration account for about one
tenth of the total anthropogenic discharge in the United Kingdom.

Table 16. *Environmental release of trace elements from municipal*
waste disposal in the United Kingdom, 1983 [103]

Disposal route	Trace elements release (t/y)		
	Cadmium	Lead	Mercury
Direct landfill	479	31 673	62.0
Incineration			
Atmosphere	5	142	5.9
Landfill	46	3 241	0.7

The landfill inputs of metals in Table 16 reveal the much larger tonnage associated with the direct disposal of MSW compared with the incinerator residues. However, the latter is likely to be of greater environmental significance at a local level because trace elements are generally present at much higher concentrations.

Environmental release of metals in Sweden

Several emission inventories have been carried out in Sweden in the last 5 years. A detailed investigation into the sources of atmospheric mercury is summarized in Table 18. These data indicate much smaller total emissions of mercury in Sweden than in the United Kingdom. In addition, refuse incineration emerges as a more important source of atmospheric mercury, mainly because of the relatively small quantity of coal combusted in Sweden. Refuse incineration accounted for over half the total released in 1985 [45]. The release of other metals from MSW incineration was less significant, accounting for only 0.3% of the total cadmium emissions.

As a result of a national programme for collecting used batteries and installing advanced flue gas treatment at some plants, the emission of mercury from waste incineration in Sweden is expected to be reduced from 3.3 t/y (Table 18) to about 1 t/y from 1987 (K. Hasselgren, personal communication, 1987).

Table 17. *Estimated atmospheric emissions of trace elements in the United Kingdom* [103]

Source	Atmospheric emission (t/y)		
	Cadmium	Lead	Mercury
Nonferrous metal production	3.7	51	5.0
Production and use of articles containing the element	ND[a]	6802	10.1
Iron and steel production	2.3	478	1.8
Fossil fuel combustion	1.9	80	25.5
Cement manufacture	1.0	36	2.5
Municipal waste incineration	5.0	142	5.9
Sewage sludge incineration	0.2	1.2	0.6
Total	14.1	7590.2	51.4

ND = not determined.
[a] Atmospheric emission not determined but probably negligible.

EMISSIONS OF HEAVY METAL AND PAH COMPOUNDS

Table 18. *Estimated emissions of mercury to air in Sweden, 1984* [45]

Source of emission	Emission of mercury kg/y
Combustion of coal and peat	200
Chlor-alkali industry	400
Crematoriums	300
Smelting (mainly copper and zinc)	900
Incineration of household waste	3300
Other	200
Total	5300

Cadmium release in the European Community

A cadmium inventory survey was carried out in 1980 for the then nine Member States of the EEC [105]. Table 19 presents a breakdown of the amounts of MSW incinerated, and the estimated discharges of cadmium to landfill and the atmosphere. Atmospheric emissions of cadmium from traditional refuse incineration were estimated to exceed 30 t in 1977; the largest contributors were, in decreasing order, the Federal Republic of Germany, France, and the United Kingdom. In comparison with other human activities and natural sources, traditional refuse incineration emerges as a major source of atmospheric release of cadmium, being responsible for about 25% of the regional total, and is the second largest single source category.

Metal release at the global level

Nriagu [109] estimates the total worldwide atmospheric emissions of several metals from human activities. Most of this information is not available at present, but Table 20 summarizes the reported emission strengths for cadmium. Traditional refuse incineration also makes a considerable input to the global release of cadmium, being the largest single source after the nonferrous metal industries.

Emissions of metals other than cadmium, mercury, and lead

In addition to the three metals covered in this report, emissions from municipal incinerators, in common with many combustion processes, will contain trace quantities of several other metals. Traditional MSW incineration may be responsible for major

Table 19. *Quantity of incinerated refuse and estimated discharges of cadmium in the European Community[a] in 1977* [105]

Country	Waste incinerated (x 10^6 t)	Cadmium in discharge medium (t)	
		Atmospheric emissions	Ash
Belgium	0.5	0.8	7.3
Denmark	2.5	3.7	33.8
France	4.9	7.4	66.0
Germany, Federal Republic of	5.4	8.1	72.9
Italy	2.6	3.9	35.1
Luxembourg	0.1	0.2	1.4
Netherlands	1.7	2.6	23.0
United Kingdom	3.2	4.8	43.0
Total	20.9	31.5	282.5

[a] Except Ireland.

Table 20. *Worldwide anthropogenic emissions of cadmium to the atmosphere* [109]

Source	Annual release (10^3 t/y)
Mining and production of nonferrous metals	4.71
Secondary nonferrous metal production	0.60
Iron and steel production	0.07
Industrial applications	0.05
Coal combustion	0.06
Waste incineration	1.40
Other sources	0.23
Total	7.12

fractions of zinc, antimony, silver, indium, and tin found in urban aerosols [17].

Some compounds of inorganic arsenic, hexavalent chromium, and nickel have caused lung cancer after occupational exposure to very high concentrations [51]. Such compounds are suspected of being potential carcinogens for the general population, even at the low concentrations found in ambient air. However, based on the emission data, the emissions of such compounds from incinerators would probably contribute very little to ambient air levels. If there is a risk for lung cancer from ambient air levels, the additional risk from incinerator emissions would not be of any practical significance.

Contribution of incinerator emissions to PAH exposure

This section considers the relative importance of MSW incineration as a source of human exposure to PAH. As the data on PAH other than BaP are severely limited, this discussion is restricted mostly to BaP.

Direct exposure via inhalation

The background concentrations of BaP quoted in the Air Quality Guidelines [51] range from less than 1 ng/m^3 in rural areas, less than 10 ng/m^3 in urban areas, and up to 40 ng/m^3 in industrial areas. The last value was recorded in the vicinity of a coke oven plant.

The concentrations of BaP reported in stack gases of incinerators range from less than 1 ng/m^3 for advanced facilities and to 60 ug/m^3 for traditional incinerators (Table 14). Maximum ground-level concentrations of pollutants from individual incinerator plant emissions can be estimated by using mathematical models. These models attempt to allow for several aspects, such as the stack height, local topography, and meteorological conditions in the given area. However, these models have yet to be validated by comparing the estimated concentrations with measurements taken in the field.

A simple dilution factor of 5×10^{-5} [24] may be applied to the stack gas emissions of incinerators to gauge the order of magnitude of ambient air concentrations of BaP at ground level. On this albeit simplistic basis, the worst case emission of BaP given in Table 14 (208 ug/m^3) for traditional incinerators would result in a maximum ambient air concentration at ground level of 10 ng/m^3. This is somewhat less than the concentration of BaP recorded in the vicinity of a coke oven plant [51]. However, for most incinerators of traditional design cited in Table 14, the maximum ambient air concentration of BaP at ground level is likely to be less than 10 ng/m^3 (Table 13), which is similar to the background concentrations in urban ambient air.

Most MSW incinerators are situated in urban areas, so the estimated maximum concentrations of PAH at ground level should be added to the prevailing background values. On the basis of the worst-case examples presented here, a poorly operated incinerator of traditional design may, at most, double the ambient air concentration of PAH. However, the impact of most traditional incinerator plants will likely be much less.

The stack emission data for advanced incinerators (Table 14) show PAH concentrations considerably less than the background concentrations in urban ambient air. This type of incinerator is therefore most unlikely to increase PAH concentrations in air close to the plant. Some recent field measurements also support this conclusion [110]. These workers recorded a maximum concentration of BaP in air at ground level of less than 1 ng/m^3, measured at two sampling sites situated 900 m and 2500 m from an incinerator. Most values recorded were less than 0.4 ng/m^3, and days with peak values of PAH in the ambient air were not correlated to days with peak PAH values in the stack gases of the incinerator plant. What type of incinerator was examined in this study is not clear, but if it were of the traditional type, it would only strengthen the conclusion that advanced incinerators will have little, if any, impact on ambient air levels of PAH.

Mutagenic activity in emissions

Besides analysis of specific chemical components in the flue gas, the biological activity may also be tested, thereby offering complementary information on potential health effects. The *Salmonella*/microsomal assay has been used to detect mutagenic activity in the emissions from different combustion sources, including MSW incineration and ambient air [111–113].

The mutagenic activity is distributed between fractions containing PAH and more polar fractions, probably containing oxygenated PAH components [113]. Thus, other substances besides unsubstituted PAH are of importance. As the identity of these compounds is not known, their potential health impact cannot be assessed.

Technical measures aimed at optimizing the combustion efficiency may reduce the emission of mutagenic substances, as well as the emission of other organic compounds. The results from mutagenicity testing can be used, for example, to compare the effect of different control devices on emissions, especially in combination with chemical analyses. However, the data generated with the *Salmonella*/microsomal assay are insufficient to estimate the mutagenic potency, and even less the carcinogenic risks of various emissions. On its own, this test can give only an insight into the potential for mutagenicity; it cannot produce quantitative data.

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Indirect exposure via deposition onto food crops

The estimated increase in the deposition rate of PAH in the vicinity of MSW incinerators (Table 13) is 0.02 to 1.5 g/ha/y. The influence of this small increase in deposition of PAH may slightly increase the PAH concentrations in food crops grown in the vicinity of a traditional incinerator, particularly one poorly situated with respect to the local geography. However, this source is not expected to increase appreciably the total daily intake of PAH.

Emissions of PAH from MSW incinerators compared with other sources

Table 21 shows that incineration and open burning of refuse contributes about 25% of the global BaP emissions to the atmosphere. The emissions from open burning of refuse and agricultural wastes will heavily outweigh the contribution from MSW incinerators, which together make up this estimate.

More recent estimates for the Nordic countries (Table 22) indicate that the contribution of MSW incinerators to emissions of PAH is probably less than 5% of the total. In Norway, aluminium smelting is the largest single source, whereas in Denmark, agricultural burning is the dominant source. If agricultural burning, forest fires, and industrial emissions are excluded, residential heating with wood contributes from 44 to 68% of the total emission in different countries. Motor vehicles is the other dominant source.

Table 21. *Estimated emissions of BaP to the atmosphere* [114]

Source	BaP in t/y		
	USA	Worldwide (excl. USA)	Worldwide
Heating and power generation	475	2 129	2 604
Industrial processes	198	847	1 045
Refuse and open burning including incinerations	588	762	1 350
Vehicles	22	23	45
Total	1 283	3 761	5 044
Contribution from incineration (%)	46	-20	27

Table 22. *Estimated emissions of PAH from different sources in the Nordic countries (kg/y)* [115]

Source	Norway	Sweden	Denmark	Finland
Residential combustion (wood, oil, gas)	49 000	105 000	29 0C0	42 000
Industrial production	230 000[a]	48 000	1 200	2 700
Power and heat generation (coal, oil, biomass)	230	6 600	550	1 100
Municipal incineration	250	2 200[b]	3 500	170
Open burning, forest fires agricultural burning	13 000	7 800	33 000	5 500
Mobile sources (gasoline and diesel automobiles)	20 000	47 000	22 000	26 000
Total	312 480	216 600	89 250	77 470
Contribution from waste incineration (%)	0.08	1.0	4.0	0.22

[a] 180 000 kg from aluminium production.
[b] Later estimated to be 840 kg/y [45].

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

1. Cadmium, lead, and mercury are the metals of prime concern because of their toxicity and the relative importance of municipal solid waste (MSW) incineration as a source of these metals emitted to the atmosphere. Considerable amounts of these metals may be emitted from incinerators of traditional design, which in turn may increase exposure among residents in the vicinity of such an incinerator.

2. As most MSW incinerators in operation are of traditional design, they are an important source of cadmium and mercury emissions to the atmosphere at a regional level and can contribute to the total human intake of these metals through the food chain. However, the contribution of lead from incinerators through this pathway is negligible in comparison with emissions from other sources, such as motor vehicles and smelters.

3. Incinerators can be designed and operated in a manner that will affect the physical and chemical properties of metallic combustion products and their distribution into bottom ash, boiler ash, fly ash, and stack emissions. Emissions of metals, such as cadmium, lead, and mercury, can be considerably reduced by use of a proper flue gas cleaning system.

4. On the basis of the limited data available, metal emissions from advanced MSW incinerators may not increase exposure levels in the population living near such facilities.

5. Advanced air pollution control systems result in increased quantities of dry or wet residues. Gas treatment residues, quench waters, and other liquid effluents require appropriate treatment prior to disposal.

6. Management of MSW incinerator residues must be done properly to prevent dispersal of ashes by fugitive emissions and contamination of surface and groundwaters due to the leachability of metals, particularly cadmium and lead.

7. The level of PAH emissions from MSW incinerators varies widely, and the factors causing these variations are not thoroughly understood. Nevertheless, tests show that advanced incinerators operating at sufficiently high temperatures and with adequate oxygen, good mixing, and adequate retention time (together reflecting good combustion conditions) result in lower formation levels of PAH than in traditional incinerators and much lower than in poorly maintained and poorly operated incinerators. The data on advanced incinerator designs indicate that highly efficient destruction of PAH can be achieved in a single-chamber incinerator.

8. The available data indicate that advanced MSW incinerators will make only a minor contribution to the PAH concentrations in ambient air.

9. At a local level, poorly operated MSW incinerators may, in the worst case, increase the ambient air concentration of PAH in urban locations by a factor of 2, and, consequently, may add to the risk of cancer induction. Furthermore, the likelihood of PAH contami-

nation of food crops in the vicinity of such incinerators cannot be excluded. However, PAH are only a fraction of the potentially toxic organic compounds emitted from incinerators.

10. At a regional level, MSW incineration makes only a small (up to a few percent) contribution to the total PAH emission and is therefore of minor importance compared with other sources.

Recommendations

1. The local environmental impact of MSW incinerators should be investigated at a few selected facilities by direct determination of cadmium, mercury, lead and PAH concentrations in air, deposited dust, soil, and vegetation. These values should be compared with appropriate background values. Studies should also be conducted at ash residue landfill sites, with particular reference to the potential contamination of surface and groundwaters by metal leachates.

2. Major sources of metals in MSW, such as cadmium, lead, and mercury, should be identified, their amounts evaluated, the significance of their removal on incineration emissions assessed, and alternative disposal routes investigated.

3. The evaluation of metal and PAH emissions should be based on a series of measurements of emissions from MSW incinerators, and the recording of corresponding operating conditions of the incinerator and flue gas cleaning devices and MSW characteristics.

4. Additional studies should be conducted to validate sampling techniques and analytical methods for the measurement of PAH and metal emissions, especially mercury, from incinerators.

5. A better and more comprehensive scientific database should be developed to refine criteria for the selection of operating conditions for traditionally designed incinerators to control more adequately PAH emissions from such facilities.

6. Many PAH are known animal carcinogens and may also be carcinogenic to humans. As the cancer risk from carcinogenic PAH is assumed to depend on dose, the emissions of these substances, even at low levels over a prolonged time, should be kept to a minimum.

7. Substituted PAH compounds emitted from MSW incinerators, such as methylated, nitrated, and oxygenated PAH, and organic compounds other than PAH, should be identified and their biological effects investigated.

8. Investigations should be continued to increase combustion efficiency that would, in turn, maximize the destruction of organic compounds and their conversion to carbon dioxide.

9. Advanced technologies of MSW incinerators and emission control systems should be implemented rapidly to reduce environmental dispersal of metals and PAH.

10. Application of genotoxicology testing in conjunction with chemical analysis should be considered as a valuable tool for assessing the impact of operating conditions and new technologies on the potential toxicity of pollutant mixtures emitted in the stack gas.

11. The location of new MSW incinerators should be carefully selected to minimize human and environmental impact from metal and PAH emissions.

12. Safe handling of ash residues, both in the incinerator plant and during transport to disposal sites, should be practised to minimize occupational exposure and fugitive emissions; assessments of the metal and PAH exposure levels of the operators should be undertaken. Liquid effluents from wet scrubbers and quenches should be properly treated prior to discharge to receiving waters.

13. The construction of small incinerators should be discouraged when adequate number of properly trained personnel cannot be provided on a cost-effective basis, and should be permitted only when proper pollution control equipment is incorporated in the incinerator facility.

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Annex 1

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Mr Milhau   Dr Viviano
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