APP/SMO/2

BILSTHORPE ENERGY CENTRE

PUBLIC INQUIRY UNDER SECTION 77 OF THE TOWN AND COUNTRY PLANNING ACT 1990 (AS AMENDED) INTO THE PROPOSED DEVELOPMENT OF AN ENERGY FROM WASTE FACILITY ON LAND AT BILSTHORPE BUSINESS PARK, BILSTHORPE, NOTTINGHAMSHIRE

PINS REFERENCE: APP/L3055/V/14/3007886
LPA REFERENCE: ES/2950

Appendices to Proof of Evidence of Stephen Othen
On
Air Quality and R1 Recovery Status

November 2015

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Case		1	2	3	4	5	6	7
		Base	Heat Export	97% power	80% waste & coke	waste, 93.7% coke	Heat plan waste	Lower CV waste
Operating hours		7,600	7,600	7,600	7,600	7,600	7,600	7,600
Energy Inputs								
Waste								
- flow	tonnes	95,000	95,000	95,000	76,000	76,000	84,220	95,000
- NCV	MJ/kg	12.581	12.581	12.581	12.581	12.581	14.46	10.5
- energy input	GJ	1,195,195	1,195,195	1,195,195	956,156	956,156	1,217,821	997,500
Coke								
- flow	tonnes	3,800	3,800	3,800	3,040	3,561	3,037	3,715
- NCV	MJ/kg	29.834	29.834	29.834	29.834	29.834	29.834	29.834
- energy input	GJ	113,369	113,369	113,369	90,695	106,227	90,621	110,833
Total Energy In while generating power	GJ	1,308,564	1,308,564	1,308,564	1,046,851	1,062,383	1,308,442	1,108,333
Auxiliary fuel								
- flow	Nm^3	54,000	54,000	54,000	54,000	54,000	54,000	54,000
- NCV	MJ/Nm^3	34.2	34.2	34.2	34.2	34.2	34.2	34.2
- energy input	GJ	1,847	1,847	1,847	1,847	1,847	1,847	1,847
- percentage used in start-up		100%	100%	100%	100%	100%	100%	100%
Power and Heat Generation								
Syngas								
- flow	kg/hr	14,949	14,949	14,949	11,959	12,137	14,948	12,662
- flow	kmol/hr	614.97	614.97	614.97	491.98	499.28	614.91	520.87
- flow	m^3/hr	14,544	14,544	14,544	11,635	11,808	14,543	12,319
- flow	m^3	110,534,400	110,534,400	110,534,400	88,426,000	89,740,800	110,526,800	93,624,400
- GCV	MJ/m^3	9.244	9.244	9.244	9.244	9.244	9.244	9.244
- NCV	MJ/m^3	8.633	8.633	8.633	8.633	8.633	8.633	8.633
- energy input	GJ	954,243	954,243	954,243	763,382	774,732	954,178	808,259
	MW	34.88	34.88	34.88	27.90	28.32	34.87	29.54
Conversion efficiency	%	72.92%	72.92%	72.92%	72.92%	72.92%	72.92%	72.93%
Gas engine efficiency	%	39.50%	39.50%	38.32%	39.50%	39.50%	39.50%	39.50%
Power generated	GJ	376,926	376,926	365,618	301,536	306,019	376,900	319,262
	MW	13.78	13.78	13.36	11.02	11.18	13.78	11.67
Gross Efficiency		28.80%	28.80%	27.94%	28.80%	28.80%	28.81%	28.81%
Electricity consumed	MW	4	4	4	4	4	4	4
	GJ	109,440	109,440	109,440	109,440	109,440	109,440	109,440
Net Efficiency		20.44%	20.44%	19.58%	18.35%	18.50%	20.44%	18.93%
Heat Exported	MW	0	4	0	0	0	0	0
	GJ	0	30,400	0	0	0	0	0
Electricity imported	kW	670	670	670	670	670	670	670
	GJ	2,798	2,798	2,798	2,798	2,798	2,798	2,798
R1 calculation								
Ew	GJ	1,195,195	1,195,195	1,195,195	956,156	956,156	1,217,821	997,500
Ep (electricity)	GJ	980,008	980,008	950,608	783,993	795,650	979,941	830,082
Ep (heat)	GJ	0	33,440	0	0	0	0	0
Ep (total)	GJ	980,008	1,013,448	950,608	783,993	795,650	979,941	830,082
Ef	GJ	113,369	113,369	113,369	90,695	106,227	90,621	110,833
Ei	GJ	9,121	9,121	9,121	9,121	9,121	9,121	9,121
R1		0.676	0.702	0.652	0.674	0.660	0.694	0.661

Briefing note

Waste (England and Wales) Regulations 2011

Version 2- April 2012

Qualifying for R1 status using the R1 energy efficiency formula

Purpose of this briefing note

To provide an overview of how, in England and Wales, an incineration plant dedicated to the processing of municipal solid waste can qualify as a recovery operation using the R1 Energy Efficiency formula in Annex II of the Waste Framework Directive 2008/98/EC (WfD).

Background to the formula

Article 3(15) of the WfD defines 'recovery' and refers to the non-exhaustive list of recovery operations in Annex II of WfD. Annex II of the WfD includes the energy efficiency formula[1] as a footnote to the R1 definition to provide an incentive for Municipal Waste Incinerators (MWI) to contribute to energy supply.

MWI operating at or above the stipulated thresholds can be classified as recovery operations for the purposes of the waste hierarchy. The threshold for plants which commenced operation prior to the end of 2008 is 0.6 and for plants which commenced operation thereafter is 0.65.

There is no requirement for MWI to achieve R1 status or have their performance assessed against the R1 formula in the Environmental Permitting Regulations 2010 (EPR). Therefore the R1 formula is only relevant for those MWI wishing to qualify as a recovery operation. The European Commission's guidelines[2] provide detailed guidance on how to interpret and apply the R1 Energy Efficiency formula.

Plants within the scope of the formula

The R1 formula applies only to those plants dedicated to the incineration of Municipal Solid Waste (MSW)[3] where the following apply:

- the plant is an incineration plant as defined by the Environmental Permitting Regulations 2010 (EPR); and
- the EPR permit (or the permit application) is for a plant processing MSW.

Plants processing a mixture of MSW and other wastes are within the scope of the formula provided that the plant is principally designed to process MSW. Where the proportion of other wastes is

1 The formula calculates the energy efficiency of the plant and expresses it as a factor which is not equivalent to a percentage or a thermal efficiency
2 The guidelines are not legally binding and could be revised as explained in the foreword
3 'municipal waste' means waste from households, as well as other waste which, because of its nature or composition, is similar to waste from households

significant the applicant will need to demonstrate that the plant is not dedicated to incinerating non-MSW and is technically capable of incinerating mixed MSW[4].

Plants processing only Solid Recovered Fuel (SRF), Refuse Derived Fuel (RDF) or similar pre-processed wastes will be considered within the scope of the formula when at least 50% of the waste being processed in the incineration plant is derived from MSW and the incineration plant is technically capable of incinerating mixed MSW[4].

Incineration plants which do not achieve recovery status will be classified as disposal operations (D10 in Annex 1 of the WfD).

Applying for R1 status

Those plants wishing to qualify as recovery operations by virtue of the R1 formula need to make an application to the Environment Agency. The application forms are available from the Waste Incineration Sector page on our website. The assessment of applications is separate from the determination of the EPR Permit and will vary as outlined in Table 1 below.

Table 1

Type of plant	Basis of assessment
A new plant applying for an EPR permit	Staged process based on: • Design information • Commissioning data • Performance data
A permitted plant which has yet to be built or commissioned	
An existing plant which is undergoing (or has undergone) a process change or contractual amendment which will influence the energy efficiency	
An existing plant which has been in operation less than 1 year	Staged process based on: • Commissioning data • Performance data
An existing plant which has been in operation for greater than 1 year	Assessment based on: • Performance data • Boiler efficiency data

4 This MSW may have undergone some pre-treatment such as sorting to remove dry recyclables

Existing plants can apply to us at any time. Operators applying for a new EPR permit or carrying out a variation of an existing EPR permit can make a R1 application at the time of the permit or variation application provided there is sufficient design information to complete the application form.

There is no fee associated with making an application but, in some cases, it may be necessary for us to instruct an independent expert to review or verify elements of the application. This is only likely to take place in stage 3 when actual plant data is being assessed and in these instances the costs will be recharged to the applicant.

The application process for new plants or modified plants

There are a maximum of 3 stages to the application process:

Stage 1 - An application based on the design information which takes into account that performance may vary over the course of a year.

Stage 2 - Provision of follow-up data following the commissioning of the plant which will include boiler efficiency data from the acceptance test.

Stage 3 - Final application based on one year's performance data.

The R1 status of new plants at the end of stages 1 and 2 is only provisional and is subject to final confirmation based on the Stage 3 assessment.

Ongoing monitoring

The operator of a qualifying plant will need to report annually to the Environment Agency on the performance of the plant as part of the annual report required by the EPR permit. The reporting period for ongoing R1 verification will be the same as that for the annual report even where the status is granted part way through the year.

A boiler efficiency derived from acceptance test is valid for five years for the purposes of an application or ongoing reporting. Thereafter the operator will need to repeat a comprehensive recalculation of the boiler efficiency at intervals not exceeding five years and provide this information to the Environment Agency.

Validity of the R1 status

The R1 status of a plant will need to be confirmed on an annual basis through the submission of the annual report. Failing to submit a report or achieve the relevant R1 threshold will generally result in the R1 status being withdrawn.

Should a plant not achieve the relevant threshold at the end of a year owing to circumstances beyond the operator's control the operator can request an extension of the R1 status while taking action to address the problems that led to the decrease in energy efficiency. The likelihood of obtaining an extension will depend on the performance of the plant over the past 3 years, the length of time it will take to address the problems and the probability of success. Plants that cannot achieve the relevant

threshold in the subsequent year will lose their R1 status. More detail is available from the Commission's guidance.

Communication of R1 status

The outcome of an application, each stage of the staged application process and the annual performance review will be communicated to the applicant in writing.

The status of plants that have applied to us for confirmation that they qualify as recovery operations by virtue of the R1 formula will also be available on our website.

Further information

Further information is available from the Waste incineration sector page on our website or the local PPC officer contactable through our customer service line below.

EUROPEAN COMMISSION
DIRECTORATE-GENERAL
ENVIRONMENT

GUIDELINES
ON THE INTERPRETATION OF THE R1 ENERGY EFFICIENCY FORMULA FOR INCINERATION FACILITIES DEDICATED TO THE PROCESSING OF MUNICIPAL SOLID WASTE ACCORDING TO ANNEX II OF DIRECTIVE 2008/98/EC ON WASTE[1]

[1] Directive 2008/98/EC on waste and repealing certain Directives – OJ L 312, 22.11.2008, p. 3.

1 Introduction

These guidelines are destined to provide legal certainty and a level playing field in the application of the energy efficiency thresholds for municipal waste incinerators in Annex II of Directive 2008/98/EC on waste (Waste Framework Directive - WFD).

The new WFD has introduced a **five-step waste hierarchy as a priority order** with waste prevention at the top followed by preparing for re-use, recycling, other recovery including energy recovery and waste disposal as the last resort. The Directive allows municipal waste incinerators to be classified as recovery operations provided they contribute to the generation of energy with high efficiency to promote the use of waste to produce energy in energy efficient municipal waste incinerators and encourage innovation in waste incineration.

In this context, it is important to note that "recovery" means any operation the principal result of which is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfil a particular function, or waste being prepared to fulfil that function, in the plant or in the wider economy (Art 3 (15) of the WFD).

The non-exhaustive list of recovery operations presented in Annex II of the WFD defines R1 as a recovery operation which is understood as "*Use principally as a fuel or other means to generate energy*". It is clarified in footnote (8) that this includes incineration facilities dedicated to the processing of municipal solid waste (MSW) only where their energy efficiency is equal to or above:

- 0.60 for installations in operation and permitted in accordance with applicable Community legislation before 1 January 2009,
- 0.65 for installations permitted after 31 December 2008,

using the following formula:

$$\text{Energy efficiency} = \frac{E_p - (E_f + E_i)}{0.97 * (E_w + E_f)}$$

In which:

***E_p** means annual energy produced as heat or electricity. It is calculated with energy in the form of electricity being multiplied by 2.6 and heat produced for commercial use multiplied by 1.1 (GJ/year)*
***E_f** means annual energy input to the system from fuels contributing to the production of steam (GJ/year)*
***E_w** means annual energy contained in the treated waste calculated using the net calorific value of the waste (GJ/year)*
***E_i** means annual energy imported excluding E_w and E_f (GJ/year)*
***0.97** is a factor accounting for energy losses due to bottom ash and radiation*

In addition, Annex II of the WFD highlights that this formula shall be applied in accordance with the Reference Document on Best Available Techniques for Waste Incineration (BREF WI).

The "R1-formula" is not strictly speaking an expression of efficiency in physics, but a performance indicator for the level of recovery of energy from waste in a plant dedicated to the incineration of municipal solid waste (MSWI). The practical impact of this provision will have to be monitored in future and the R1 formula may be revised in 2014 in accordance with the provisions of article 37(4) of the WFD, and if necessary to keep it up to date with technological progress.

For historical development of the formula and its link to the Integrated Pollution Prevention and Control Reference Document on the Best Available Techniques for Waste Incineration from August 2006 (BREF WI) see Annex 1.

For better readability, this document specifies major topics in specific thematic areas in shaded boxes and summarises the major elements of guidance in boxes at the end of each chapter.

It should be noted that this guidance only reflects the opinion of the Commission services and is not legally binding. A final binding legal interpretation of EU legislation can only be provided by the Court of Justice of the European Union. This guidance is without prejudice to the position the Commission might take should related issues arise in a procedure before the Court of Justice.

1.1 Scope of the Energy Efficiency Formula

Annex II, footnote (*) of the WFD clearly restricts the scope of the formula to "incineration facilities dedicated to the processing of municipal solid waste" (MSWI). The WFD should, pursuant to its recital 20, clarify when incineration of (MSW) is energy-efficient and may be considered as recovery operation.

Waste incinerators dedicated to the incineration of municipal waste are waste incinerators which have the permit and are technically designed in a way so that they are capable to incinerate mixed municipal solid waste.

The R1 formula does not apply to co-incineration plants and facilities dedicated to the incineration of hazardous waste, hospital waste, sewage sludge or industrial waste.

Installations shall correspond to the IPPC activity 5.2. "Installations for the incineration of municipal waste (household waste and similar commercial, industrial and institutional wastes) with a capacity exceeding 3 tonnes per hour" (it should be noted that the capacity limit in this context is not applicable in the context of the R1 formula). However, this activity description will change under the IED, Annex I, as indicated below:

> 5.2 Disposal or recovery of waste in waste incineration plants or in waste co-incineration plans:
>
> (a) for non-hazardous waste with a capacity exceeding 3 tonnes per hour;
>
> (b) for hazardous waste with a capacity exceeding 10 tonnes per day.

In the context of IED, installations dedicated to the incineration of municipal waste shall correspond to a sub-sector of activity 5.2 recognizing that: (1) only if the facility is dedicated to the incineration of municipal solid waste will it fall within the R1 energy efficiency thresholds of the WFD and (2) that the R1-formula does not apply to co-incinerators.

Municipal waste is classified in chapter 20 of Commission Decision 2000/532/EC on the list of waste. Usually, MSWI are installations permitted for the incineration of 'mixed municipal waste'. Mixed municipal waste is defined in Art 3(3) WID as waste from households as well as commercial, industrial and institutional waste, which because of its nature and composition is similar to waste from households, excluding separately collected fractions of recyclable waste.

In addition, other waste streams can be accepted by MSWI if listed in the permit for the IPPC category 5.2, if applicable, or the permit according to WID. Authorization of any waste input, except for mixed municipal solid waste, shall be in line with the BREF on waste incineration and with the waste hierarchy (Art 4 WFD).

In practice, the waste input into a MSWI is made of different mixed and heterogeneous fractions which are blended before feeding the hopper in order to optimize the combustion process.

The calculation of the R1 formula shall be done on the waste composition which is actually incinerated in a facility, not only on the part of the waste which is classified as municipal waste or mixed municipal waste.

In case an incineration plant has two separate lines (one for hazardous waste and one for MSW), only the line for MSW can apply for the R1 status according to the formula.

Non-municipal wastes can be accepted as long as specified in the permit in accordance with the IPPC and WID and the BREF document, although primarily other treatment options might be preferred. Separately collected waste fractions should be managed in line with the waste hierarchy.

The calculation of the E_w as a parameter for the R1 efficiency is based on the actual waste mix incinerated.

1.2 Principles of self-sufficiency and proximity and the waste hierarchy

Together with the introduction of the R1 formula, the principles of self-sufficiency and proximity have been extended from waste disposal installations to the recovery of mixed municipal waste collected from private households, including where such collection also covers such waste from other producers.

The fact that municipal waste treated in an R1-facility is to be regarded as recovered has to be distinguished from the question of whether the recovery of a certain waste in such a facility is to be seen as a waste management option with the best environmental outcome considering the waste hierarchy and taking into account life-cycle thinking (Art 4 WFD). Certain waste streams like paper, glass, plastic, and metals can be used with higher resource efficiency when they are separately collected from other municipal wastes and recycled.

DIRECTIVES

DIRECTIVE 2010/75/EU OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL

of 24 November 2010

on industrial emissions (integrated pollution prevention and control)

(Recast)

(Text with EEA relevance)

THE EUROPEAN PARLIAMENT AND THE COUNCIL OF THE EUROPEAN UNION,

Having regard to the Treaty on the Functioning of the European Union, and in particular Article 192(1) thereof,

Having regard to the proposal from the European Commission,

Having regard to the opinion of the European Economic and Social Committee ([1]),

Having regard to the opinion of the Committee of the Regions ([2]),

Acting in accordance with the ordinary legislative procedure ([3]),

Whereas:

(1) A number of substantial changes are to be made to Council Directive 78/176/EEC of 20 February 1978 on waste from the titanium dioxide industry ([4]), Council Directive 82/883/EEC of 3 December 1982 on procedures for the surveillance and monitoring of environments concerned by waste from the titanium dioxide industry ([5]), Council Directive 92/112/EEC of 15 December 1992 on procedures for harmonising the programmes for the reduction and eventual elimination of pollution caused by waste from the titanium dioxide industry ([6]), Council Directive 1999/13/EC of 11 March 1999 on the limitation of emissions of volatile organic compounds due to the use of organic solvents in certain activities and installations ([7]), Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste ([8]), Directive 2001/80/EC of the European Parliament and of the Council of 23 October 2001 on the limitation of emissions of certain pollutants into the air from large combustion plants ([9]) and Directive 2008/1/EC of the European Parliament and of the Council of 15 January 2008 concerning integrated pollution prevention and control ([10]). In the interests of clarity, those Directives should be recast.

(2) In order to prevent, reduce and as far as possible eliminate pollution arising from industrial activities in compliance with the 'polluter pays' principle and the principle of pollution prevention, it is necessary to establish a general framework for the control of the main industrial activities, giving priority to intervention at source, ensuring prudent management of natural resources and taking into account, when necessary, the economic situation and specific local characteristics of the place in which the industrial activity is taking place.

(3) Different approaches to controlling emissions into air, water or soil separately may encourage the shifting of pollution from one environmental medium to another rather than protecting the environment as a whole. It is, therefore, appropriate to provide for an integrated approach to prevention and control of emissions into air, water and soil, to waste management, to energy efficiency and to accident prevention. Such an approach will also contribute to the achievement of a level playing field in the Union by aligning environmental performance requirements for industrial installations.

([1]) OJ C 182, 4.8.2009, p. 46.

([2]) OJ C 325, 19.12.2008, p. 60.

([3]) Position of the European Parliament of 10 March 2009 (OJ C 87 E, 1.4.2010, p. 191) and position of the Council at first reading of 15 February 2010 (OJ C 107 E, 27.4.2010, p. 1). Position of the European Parliament of 7 July 2010 (not yet published in the Official Journal) and decision of the Council of 8 November 2010.

([4]) OJ L 54, 25.2.1978, p. 19.

([5]) OJ L 378, 31.12.1982, p. 1.

([6]) OJ L 409, 31.12.1992, p. 11.

([7]) OJ L 85, 29.3.1999, p. 1.

([8]) OJ L 332, 28.12.2000, p. 91.

([9]) OJ L 309, 27.11.2001, p. 1.

([10]) OJ L 24, 29.1.2008, p. 8.

2. In the case of multi-fuel firing combustion plants covered by Article 30(2), which use the distillation and conversion residues from the refining of crude-oil for own consumption, alone or with other fuels, the following emission limit values may be applied instead of the emission limit values set according to paragraph 1:

(a) where, during the operation of the combustion plant, the proportion contributed by the determinative fuel to the sum of the thermal inputs delivered by all fuels is 50 % or more, the emission limit value set in Part 1 of Annex V for the determinative fuel;

(b) where the proportion contributed by the determinative fuel to the sum of the thermal inputs delivered by all fuels is less than 50 %, the emission limit value determined in accordance with the following steps:

(i) taking the emission limit values set out in Part 1 of Annex V for each of the fuels used, corresponding to the total rated thermal input of the combustion plant;

(ii) calculating the emission limit value of the determinative fuel by multiplying the emission limit value, determined for that fuel according to point (i), by a factor of two, and subtracting from this product the emission limit value of the fuel used with the lowest emission limit value as set out in Part 1 of Annex V, corresponding to the total rated thermal input of the combustion plant;

(iii) determining the fuel-weighted emission limit value for each fuel used by multiplying the emission limit value determined under points (i) and (ii) by the thermal input of the fuel concerned and by dividing the product of this multiplication by the sum of the thermal inputs delivered by all fuels;

(iv) aggregating the fuel-weighted emission limit values determined under point (iii).

3. In the case of multi-fuel firing combustion plants covered by Article 30(2), which use the distillation and conversion residues from the refining of crude-oil for own consumption, alone or with other fuels, the average emission limit values for sulphur dioxide set out in Part 7 of Annex V may be applied instead of the emission limit values set according to paragraphs 1 or 2 of this Article.

Article 41

Implementing rules

Implementing rules shall be established concerning:

(a) the determination of the start-up and shut-down periods referred to in point 27 of Article 3 and in point 1 of Part 4 of Annex V; and

(b) the transitional national plans referred to in Article 32 and, in particular, the setting of emission ceilings and related monitoring and reporting.

Those implementing rules shall be adopted in accordance with the regulatory procedure referred to in Article 75(2). The Commission shall make appropriate proposals not later than 7 July 2011.

CHAPTER IV

SPECIAL PROVISIONS FOR WASTE INCINERATION PLANTS AND WASTE CO-INCINERATION PLANTS

Article 42

Scope

1. This Chapter shall apply to waste incineration plants and waste co-incineration plants which incinerate or co-incinerate solid or liquid waste.

This Chapter shall not apply to gasification or pyrolysis plants, if the gases resulting from this thermal treatment of waste are purified to such an extent that they are no longer a waste prior to their incineration and they can cause emissions no higher than those resulting from the burning of natural gas.

For the purposes of this Chapter, waste incineration plants and waste co-incineration plants shall include all incineration lines or co-incineration lines, waste reception, storage, on site pretreatment facilities, waste-, fuel- and air-supply systems, boilers, facilities for the treatment of waste gases, on-site facilities for treatment or storage of residues and waste water, stacks, devices and systems for controlling incineration or co-incineration operations, recording and monitoring incineration or co-incineration conditions.

If processes other than oxidation, such as pyrolysis, gasification or plasma process, are applied for the thermal treatment of waste, the waste incineration plant or waste co-incineration plant shall include both the thermal treatment process and the subsequent incineration process.

If waste co-incineration takes place in such a way that the main purpose of the plant is not the generation of energy or production of material products but rather the thermal treatment of waste, the plant shall be regarded as a waste incineration plant.

2. This Chapter shall not apply to the following plants:

(a) plants treating only the following wastes:

(i) waste listed in point (b) of point 31 of Article 3;

(ii) radioactive waste;

(iii) animal carcasses as regulated by Regulation (EC) No 1774/2002 of the European Parliament and of the Council of 3 October 2002 laying down health rules concerning animal by-products not intended for human consumption [1];

[1] OJ L 273, 10.10.2002, p. 1.

environmental protection uk

IAQM Institute of Air Quality Management

Land-Use Planning & Development Control: Planning For Air Quality

Guidance from Environmental Protection UK and the Institute of Air Quality Management for the consideration of air quality within the land-use planning and development control processes.

May 2015 (v1.1)

Contents

Acknowledgements

Disclaimer: This guidance was produced as a result of the voluntary contribution of individuals within the Working Group, who are members of EPUK and/or IAQM, for which both organisations are grateful. Whilst this guidance represents a consensus view of the Working Group, it does not necessarily represent the view of individual members.

The information in this document is intended to provide guidance for those working in land-use planning and development control, and does not constitute legal advice. The IAQM and EPUK have endeavoured to ensure that all information in this document is accurate. However, neither organisation will accept liability for any loss, damage or inconvenience arising as a consequence of any use of or the inability to use any information in this guidance. We are not responsible for any claims brought by third parties arising from your use of this Guidance.

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Environmental Protection UK: Environmental Protection UK is a national charity that provides expert policy analysis and advice on air quality, land quality, waste and noise and their effects on people and communities in terms of a wide range of issues including public health, planning, transport, energy and climate. Membership of Environmental Protection UK is drawn from local authorities, industry, consultancies and individuals who are practicing professionals in their field, or have an interest in the environment.

The Institute of Air Quality Management (IAQM): IAQM aims to be the authoritative voice for air quality by maintaining, enhancing and promoting the highest standards of working practices in the field and for the professional development of those who undertake this work. Membership of IAQM is mainly drawn from practising air quality professionals working within the fields of air quality science, air quality assessment and air quality management.

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Document revision
v1.1, 09/06/15 – Table 6.2, row 7: 'kWh' corrected to 'kW

1. Purpose and structure of this guidance

1.1 Environmental Protection UK (EPUK) and the Institute of Air Quality Management (IAQM) have produced this guidance to ensure that air quality is adequately considered in the land-use planning and development control processes.

1.2 The spatial planning system has an important role to play in improving air quality and reducing exposure to air pollution. Both the development of local planning policies and the determination of individual planning applications are important, the former setting the framework for the latter. This guidance focuses on development control, but also stresses the importance of having good air quality policies within local authority planning frameworks.

1.3 The intended audience for this guidance is made up of air quality and planning officers within local authorities, and developers and consultants involved in the preparation of development proposals and planning applications.

1.4 This document has been developed for professionals operating within the planning system. It provides them with a means of reaching sound decisions, having regard to the air quality implications of development proposals. It also is anticipated that developers will be better able to understand what will make a proposal more likely to succeed. This guidance, of itself, can have no formal or legal status and is not intended to replace other guidance. For example, industrial development regulated by the Environment Agency, and requiring an Environmental Permit, is subject to the Horizontal Guidance Note H1[1], while for major new road schemes, Highways England has prepared a series of advice notes on assessing impacts and risk of non-compliance with limit values[2].

1.5 This guidance document is particularly applicable to assessing the effect of changes in exposure of members of the public resulting from residential and mixed-use developments, especially those within urban areas where air quality is poorer. It will also be relevant to any other forms of development where a proposal could affect local air quality and for which no other guidance exists. This guidance is not intended to be applied to the assessment of air quality impacts on designated nature conservation sites[3].

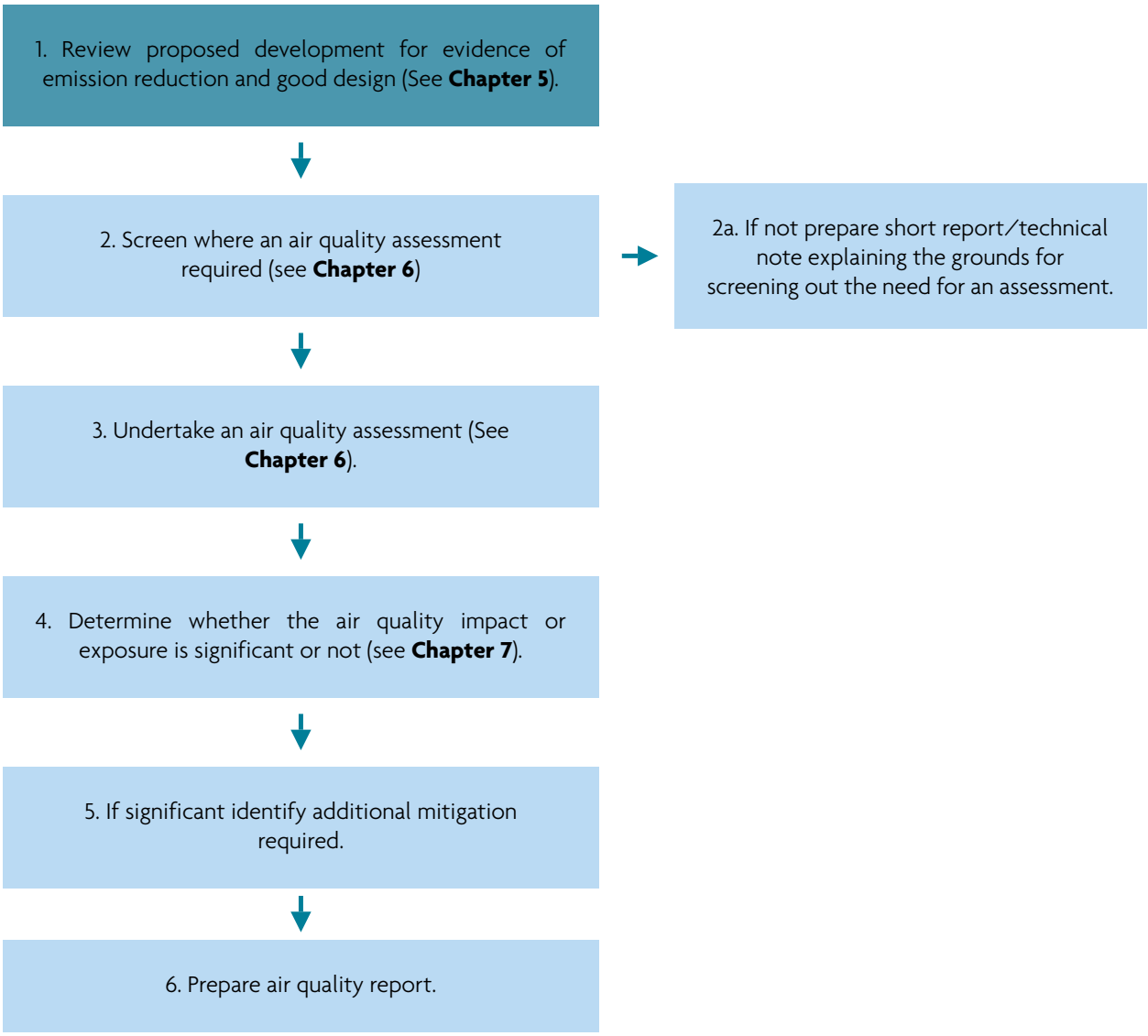
1.6 The guidance sets out why air quality is an important consideration in many aspects of local authority spatial planning. It emphasises how good spatial planning can reduce exposure to air pollution, as well as providing other benefits of well-being to the wider community. It also emphasises the importance of applying good design and 'best-practice'[4] measures to **all developments**, to reduce both pollutant emissions and human exposure. It also provides guidance on how air quality considerations of individual schemes may be considered within the development control process, by suggesting a framework for the assessment of the impacts of developments on local air quality.

1.7 Chapters 1 to 4 of this guidance set out the role of the planning regime, the important links between air quality and human health, and the links between planning and environmental assessment. Chapters 5 to 8 then describe the roles of the local authority and developer/applicant in the process through which air quality and planning decisions are taken. More specifically, Chapter 5 deals with the overarching concepts of land-use planning and air quality that should be applied throughout the strategic planning and development control processes; it emphasises that **all developments** should incorporate good principles of design with regard to minimising emissions and the reduction of impacts on local air quality. Chapters 6 to 8 then deal with the assessment of individual planning applications; the approach set out herein is founded on the concept that the principles set out in Chapter 5 are firmly adhered to, but recognises that within the development control process decisions have to be made by local planning authorities on a case-by-case basis. A flow chart describing the overall process through Chapters 5 to 8 is shown below in **Figure 1**.

1.8 This guidance is not intended to cover the specific assessment of odour or construction dust effects that some developments may give rise to. Separate guidance has been published by IAQM i.e. *'Guidance on the assessment of odour for planning'* and *'Guidance on the assessment of dust from demolition and construction'* and these guidance documents should be consulted as appropriate[5].

1.9 This guidance document applies to the planning system in England and Wales. It is intended that a separate document will be adapted for use in the Scottish and/or Northern Ireland planning systems. Meanwhile, it is considered that the general principles of air quality assessment set out herein are applicable in all parts of the United Kingdom.

Figure 1: Procedure for Evaluating New Developments



[1] www.gov.uk/government/uploads/system/uploads/attachment_data/file/298239/geho0410bsil-e-e.pdf.
[2] www.standardsforhighways.co.uk/ians.
[3] The IAQM and the Chartered Institute of Ecology and Environmental Management are considering (as of Spring 2015) where such guidance would be useful for professionals working in this area.
[4] Best practice in this guidance implies those measures which are currently considered to be the best available – this does not preclude better practice in the future.
[5] http://iaqm.co.uk/guidance.

6. Undertaking an Air Quality Assessment

Purpose

6.1 The air quality assessment is undertaken to inform the decision making with regard to the development. It does not, of itself, provide a reason for granting or refusing planning permission. Almost all development will be associated with new emissions if the development is considered in isolation. In most cases, therefore, development will be associated with adverse impacts. These impacts require quantification and evaluation in the context of air quality objectives and existing air quality. The significance of the effects arising from the impacts on air quality will depend on a number of factors and will need to be considered alongside the benefits of the development in question. Development under current planning policy is required to be sustainable and the definition of this includes social and economic dimensions, as well as environmental. Development brings opportunities for reducing emissions at a wider level through the use of more efficient technologies and better designed buildings, which could well displace emissions elsewhere, even if they increase at the development site. Conversely, development can also have adverse consequences for air quality at a wider level through its effects on trip generation.

6.2 Where a development requires an air quality assessment, this should be undertaken using an approach that is robust and appropriate to the scale of the likely impacts. One key principle is that the assessment should be transparent and thus, where reasonable, all input data used, assumptions made, and the methods applied should be detailed in the report (or appendices).

6.3 As set out in the introduction in **Chapter 1**, this guidance document is not intended to *replace* guidance that exists for certain types of development, notably:

- industrial developments that require a Permit;
- highways schemes promoted by Highways England; or
- activities associated with sources of dust (e.g. mineral extraction, waste handling, construction) or odours.

Separate guidance is available for these sources. Clearly, where new developments are located in the vicinity of such sources, the potential impacts of their operation on the proposed development will need to be considered. This should make use of the guidance for these other sources, adapted as necessary using professional judgement.

6.4 The matter of industrial development and its regulation by the Environment Agency, Natural Resources Wales or a local authority deserves some further consideration in a planning context. The 'H1' guidance provided by the Environment Agency alluded to above is intended (in part) to assist in the determination of Best Available Techniques for an installation regulated under IPPC[28]. This guidance document has been written so as to be complementary to H1 and not a substitute for it. The H1 methodology has not been designed for conducting an assessment to accompany a planning application, especially one undertaken for the EIA Regulations. In these circumstances, a framework is required that allows the assessor to describe the degree of impacts before reaching a conclusion on significance of the effects. The H1 methodology provides some useful elements of such a framework and, where relevant, these have been used in this guidance, partly for reasons of consistency. It must be recognised, however, that the H1 assessment methodology and the assessment guidance in this document serve different purposes. The former is intended for the purpose of screening out insignificant emissions of individual pollutants and identifying where there is a *risk* of a potentially significant effect on the environment through the release of some pollutants, as part of the impact assessment in support of an application under the Environmental Permitting Regulations. The latter is intended to provide a means of reaching a conclusion on whether the proposed development has a likely significant effect on local air quality, taking into account the overall degree of the impacts and other factors as appropriate. In each case, the term 'significant effect' has a deliberately different meaning and context.

This document is not intended to address impacts on nature conservation sites, for which a different form of assessment is required.

The need for an air quality assessment

6.5 It is established good practice to consult with the Local Planning Authority (and/or its air quality specialists) to gain agreement on the need for an air quality assessment in support of a planning application and if one is required, the approach and methodology that will be used. The Planning Practice Guidance at paragraph 6 makes this point. There is however a prior step in the consultation process, which is to determine the very need for an assessment. If an assessment is required, the approach and methodology can then be constructed to deal with the key issues driving the need for the assessment.

6.6 To inform the consultation process, it will be important to identify the locations of any AQMAs relative to the proposed development, the main existing and proposed sources of atmospheric pollution and the location of existing and proposed human-health sensitive receptors.

6.7 It is reasonable to expect that an assessment will be required where there is the risk of a significant air quality effect, either from a new development causing an air quality impact or creating exposure to

high concentrations of pollutants for new residents. To a large extent, professional judgement will be required to determine whether an air quality assessment is necessary as it is not possible to apply an exact and precise set of threshold criteria to cover the wide variety of development proposals. The following tables provide criteria that may be useful to guide the consultation process in establishing the need for an assessment. They separately consider:

- the impacts of existing sources in the local area on the development; and
- the impacts of the development on the local area.

6.8 Where an air quality assessment is identified as being required, this may be either a Simple or a Detailed Assessment. A Simple Assessment is one relying on already published information and without quantification of impacts, in contrast to a Detailed Assessment that is completed with the aid of a predictive technique, such as a dispersion model. Much of the discussion in this Section relates to Detailed Assessments.

6.9 The criteria provided are precautionary and should be treated as indicative; in some instances it may be appropriate to amend them on the basis of professional judgement.

Impacts of the Local Area on the Development

6.10 There may be a requirement to carry out an air quality assessment for the impacts of the local area's emissions on the proposed development itself, to assess the exposure that residents or users might experience. This will need to be a matter of judgement and should take into account:

- the background and future baseline air quality and whether this will be likely to approach or exceed the values set by air quality objectives;
- the presence and location of Air Quality Management Areas as an indicator of local hotspots where the air quality objectives may be exceeded;
- the presence of a heavily trafficked road, with emissions that could give rise to sufficiently high concentrations of pollutants (in particular NO_2), that would cause unacceptably high exposure for users of the new development; and
- the presence of a source of odour and/or dust that may affect amenity for future occupants of the development.

Impacts of the Development on the Local Area

6.11 In the case of an assessment of the impacts of a development in the local area, a two-stage approach is suggested. The **first stage** is intended to screen out smaller development and/or developments where impacts can be considered to have insignificant effects[29]. The **second stage** relates to specific details regarding the proposed development and the likelihood of air quality impacts.

6.12 **Stage 1** requires any of the criteria under (A) coupled with any of the criteria under (B) in **Table 6.1** to apply before it is considered appropriate to proceed to Stage 2. If none of the criteria are met then there should be no requirement to carry out an air quality assessment for the impact of the proposed development on the local area, and the impacts can be considered to have insignificant effects.
Table 6.1 sets out the Stage 1 criteria designed to remove the need to assess impacts arising from small developments.

Table 6.1: Stage 1 Criteria

Criteria to Proceed to Stage 2
A. If any of the following apply: • 10 or more residential units or a site area of more than 0.5ha • more than 1,000 m^2 of floor space for all other uses or a site area greater than 1ha
B. Coupled with any of the following: • the development has more than 10 parking spaces • the development will have a centralised energy facility or other centralised combustion process
Note: Consideration should still be given to the potential impacts of neighbouring sources on the site, even if an assessment of impacts of the development on the surrounding area is screened out.

6.13 The criteria in **Table 6.2** provide more specific guidance as to when an air quality assessment is likely to be required to assess the impacts of the proposed development on the local area. The criteria are more stringent where the traffic impacts may arise on roads where concentrations are close to the objective. The presence of an AQMA is taken to indicate the possibility of being close to the objective,

Table 6.2: Indicative criteria for requiring an air quality assessment

The development will:	Indicative Criteria to Proceed to an Air Quality Assessment [a]
1. Cause a significant change in Light Duty Vehicle (LDV) traffic flows on local roads with relevant receptors. (LDV = cars and small vans <3.5t gross vehicle weight)	A change of LDV flows of: - more than 100 AADT within or adjacent to an AQMA - more than 500 AADT elsewhere
2. Cause a significant change in Heavy Duty Vehicle (HDV) flows on local roads with relevant receptors. (HDV = goods vehicles + buses >3.5t gross vehicle weight)	A change of HDV flows of - more than 25 AADT within or adjacent to an AQMA - more than 100 AADT elsewhere
3. Realign roads, i.e. changing the proximity of receptors to traffic lanes.	Where the change is 5m or more and the road is within an AQMA
4. Introduce a new junction or remove an existing junction near to relevant receptors.	Applies to junctions that cause traffic to significantly change vehicle accelerate/decelerate, e.g. traffic lights, or roundabouts.
5. Introduce or change a bus station.	Where bus flows will change by: - more than 25 AADT within or adjacent to an AQMA - more than 100 AADT elsewhere
6. Have an underground car park with extraction system.	The ventilation extract for the car park will be within 20 m of a relevant receptor Coupled with the car park having more than 100 movements per day (total in and out)
7. Have one or more substantial combustion processes	Where the combustion unit is: - any centralised plant using bio fuel - any combustion plant with single or combined thermal input >300kW - a standby emergency generator associated with a centralised energy centre (if likely to be tested/used >18 hours a year)
8. Have a combustion process of any size	Where the pollutants are exhausted from a vent or stack in a location and at a height that may give rise to impacts at receptors through insufficient dispersion. This criterion is intended to address those situations where a new development may be close to other buildings that could be residential and/or which could adversely affect the plume's dispersion by way of their size and/or height.

but where whole authority AQMAs are present and it is known that the affected roads have concentrations below 90% of the objective, the less stringent criteria are likely to be more appropriate.

6.14 Where an air quality assessment is identified as being required, then this may take the form of either a Simple Assessment or a Detailed Assessment (see paragraph 6.6 for more details). In other words, passing a screening criterion in **Table 6.2** does not automatically lead to the requirement for a Detailed Assessment.

6.15 If none of the criteria are met, then there should be no requirement to carry out an air quality assessment for the impact of the development on the local area, and the impacts can be considered as having an insignificant effect. This should be agreed with the local planning authority. It may still be necessary to carry out calculations of emissions, as required by some location authorities (e.g. the GLA's SPG on Sustainable Design and Construction (2014) or the Sussex Air Quality Partnership's Air Quality and Emissions Mitigation Guidance for Sussex Authorities (2013) updated January 2014).

Content of an air quality assessment

6.16 The intent of an air quality assessment is to demonstrate the likely changes in air quality or exposure to air pollution, as a result of a proposed development. Often these changes will be quantified, although in some instances a qualitative assessment will be sufficient. Ultimately, the planning authority has to use this information to form its own view on the "significance" of the effects of air quality impacts, and thereby the priority given to air quality concerns in determining the application. The assessment therefore needs to provide sufficient information to allow this decision to be made.

6.17 In some circumstances, there will be an existing permission for development on the site that has not yet been exercised. In the planning system, the estimated emissions from the existing permission could be considered as part of the future baseline and thus a revised application for the site would give rise to an incremental change emission from that associated with the extant permission. This guidance recommends that impacts be assessed for the new permission sought against the current baseline for the site, disregarding the extant permission; this will reflect the 'real world' increase experienced by receptors.

6.18 It is important that an agreement is reached between the applicant and the local authority as to the proposed assessment methodology. The basis of the assessment should be to compare the air quality following completion of the development with that expected at that time without the development (the future 'baseline'). Comparison with existing conditions will also be required, as current conditions are those with which people are familiar. There are three basic steps in an assessment:

i. Assess the existing air quality in the study area (existing baseline);

ii. Predict the future air quality without the development in place (future baseline which may or may not include the contribution of committed development);

iii. Predict the future air quality with the development in place (with development).

6.19 The possibility of cumulative impacts should also be considered. Therefore, there may be a case for modelling another future scenario, with committed development excluded, to allow the cumulative impact of all such future developments with planning permission to be assessed as one combined impact at selected receptors. In most circumstances it is more likely that committed development would be included in the future baseline where the information exists to facilitate this. It is difficult to include other planning applications yet to be determined, as the outcome is not certain.

6.20 The report prepared detailing the results of the assessment should contain the following information (but not necessarily in this order):

a. *Relevant details of the proposed development*. A description containing information relevant to the air quality assessment should be provided, although a fully detailed description of the proposal is not required. This should identify any on-site sources of pollution and an overview of the expected traffic changes or the changes in emissions from the site for a specified year, e.g. the opening year or year the project is completed if phased. A brief introduction to the sensitivity of the area should also be provided, noting the presence of an AQMA and any nearby sources that may affect the local air quality. The proposed location of any sensitive receptors in relation to these nearby sources should be described. An introduction to the pollutants and sources to be assessed should be provided and, if appropriate, those that have been scoped out of further assessment.

b. *The policy context for the assessment*. This should summarise the national and local policies that should be taken into account in the assessment. In London this will also include the Mayor's policies. This is especially important where there are local policies designed to improve air quality.

c. *Description of the relevant air quality standards and objectives*. Most air quality assessments will be carried out to assess compliance with UK air quality objectives.

d. *The basis for determining significance of effects arising from the impacts*. The descriptors used for describing the severity of impacts should be set out, together with the basis for determining the significance of the effects arising from air quality impacts.

e. *Details of the assessment methods*. This section should provide details of the methods, including the model (and version number) and the input data used for the assessment and any assumptions that have been made. Where a commonly applied method is used, a detailed description of the model itself is not required. Details should be provided on all local input data and assumptions, including:

- the emission data and their source, with details where non-standard data are used;
- source of the meteorological data, with a description of how representative they are of the conditions in the vicinity of the proposed development;
- baseline pollutant concentrations;
- background pollutant concentrations;
- choice of baseline year;
- basis for NO_x:NO_2 calculations.

There will be some variation between requirements for reporting data relating to point sources and road traffic. The former will have some physical properties of the emission to be reported, i.e. stack height, diameter, emission velocity and exit temperature. The latter will require details of assumptions made regarding emission factors and features of the traffic flows used in the model, such as speeds and vehicles types, e.g. percentage of heavy duty vehicles (HDVs) in the traffic.

f. *Model verification*. This will normally be expected for modelling of road traffic emissions, but is not practicable for point-source modelling. If verification is not done, then some justification or explanation will be required. Model verification involves a comparison of the predicted versus measured concentrations, and allows an adjustment to be made to account for systematic errors. Such errors may include uncertainties in traffic flow, vehicle emission factors and estimated background concentrations, as well as limitations of the model to represent dispersion in settings where air flow is affected by roadside buildings, trees etc.. Model verification will be important, especially where predicted concentrations are close to the objective, and should be based on the most appropriate available monitoring data (and for some schemes it may be necessary to carry out specific monitoring to allow robust model verification to be undertaken). A more complete description of the approach to model verification is provided in LAQM.TG(09)[30]. Full details of the verification should be provided in the assessment.

g. *Identification of sensitive locations*. Local receptors should be identified, including residential and other properties close to and within the proposed development, as well as alongside roads significantly affected by the development, even if well away from the development site, and especially if within AQMAs. These receptors will represent locations where people are likely to be exposed for the appropriate averaging time (dependent on the air quality objective being assessed against).

h. *Description of baseline conditions*. The findings of any site visit(s) and/or desktop investigations will be set out, noting sources that may affect local air quality. A description of available monitoring data will be important to help define baseline conditions and put the model results into context. Where monitoring data are included in the report, it will be important to include details of the monitoring locations, the monitoring method, sampling period, data capture and any adjustments applied to the data, such as diffusion tube bias adjustment factors. Reference should also be made to the background maps produced by Defra, together with any adjustments of these mapped values to take account of local monitoring (but only where the monitoring is at true background sites). Reference should also be made to the Defra maps showing sections of road where the limit value is exceeded, as these represent the 'official' exceedences of the limit value, as reported to the European Commission. These maps are only available (at the time of writing) for 2013 and not for any future years.

i. *Assessment of impacts*. Results of modelling the 'with development' scenario should be clearly set out in tables, and where appropriate as concentration contours on maps of the study area. Comparisons should be made with the 'no development' conditions. Differences in concentrations between 'with development' and 'no development' conditions should also be tabulated. Descriptions of the impacts at the individual receptors should be provided (see section below), taking into account the absolute concentrations in relation to the air quality objectives. A comment on the sensitivity of the results to input choices is desirable, so that a view make be taken of the uncertainties.

j. *Description of construction phase impacts*. These impacts will relate primarily to dust emissions, which give rise to dust soiling and elevated PM_{10} concentrations, although construction plant and vehicles may need assessment. The assessment should take into consideration the likely activities, duration and mitigation measures to be implemented. The distance over which impacts are likely to occur and an estimate of the number of properties likely to be affected should be included. This assessment should follow the guidance set out by the IAQM[31].

k. *Cumulative impacts and effects*. In many cases, the impact of the development being assessed will have a cumulative effect with other planned developments, which may or may not have planning permission. Where these developments have been granted planning consent and are therefore 'committed' developments, their impacts should be assessed cumulatively with those of the application site. The contribution of these committed developments should be accounted for in the 'future baseline', provided that their contributions can be quantified. This situation can arise when several such developments are contributing additional road traffic on one stretch of road. In some particular cases, there may be another notable proposed development (without planning permission) in close proximity that could contribute an impact at receptors in combination with the primary development being assessed. In these circumstances, itmay be necessary to quantify this combined impact for selected

receptors and assess it against the future baseline. These occasions and the need for this form of scenario assessment will be rare.

l. *Mitigation measures*. In those cases where a significant effect is identified then the measures to be employed to avoid, reduce and, where appropriate, offset this effect should be set out. **Even where the effect is judged to be insignificant, consideration should be given to the application of good design and good practice measures, as outlined in Chapter 5**. This is especially the case for developments that increase emissions of particulate matter, as all reductions in emissions will be beneficial for human health.

m. *Summary of the assessment results.* This should include:

- Impacts during the construction phase of the development (usually on dust soiling and PM_{10} concentrations);
- Impacts on existing receptors during operation (usually on concentrations of nitrogen dioxide, PM_{10} and $PM_{2.5}$);
- Impacts of existing sources on new receptors, particularly where new receptors are being introduced into an area of high pollution;
- Any exceedances of the air quality objectives arising as a result of the development, or any worsening of a current breach (including the geographical extent);
- Whether the development will compromise or render inoperative the measures within an Air Quality Action Plan, where the development affects an AQMA;
- The significance of the effect of any impacts identified; and
- Any apparent conflicts with planning policy.

6.21 Most assessments are carried out for the first year of the proposed development's use, as this will generally represent the worst-case scenario. This is because background concentrations are predicted to decline in future years, as emissions from new vehicles are reduced by the progressive introduction of higher emissions standards. Where development is phased, however, it may also be appropriate to assess conditions for the opening years of each new phase.

Agreement of datasets and methodologies

6.22 Before undertaking an assessment, every effort should be made to obtain agreement between the planning authority and the assessor on the appropriate datasets and methodologies to be used, as described above.

6.23 It is important to recognise that the focus of the procedures used by local authorities to prepare their LAQM reports is designed specifically for the purpose of identifying whether any air quality objectives are likely to be exceeded. An air quality assessment for a development will need to go beyond this, with attention given to defining the magnitude of the changes that will take place, even where objectives are not exceeded. Nevertheless, the technical guidance to help local authorities carry out their LAQM duties includes some useful information on carrying out an air quality assessment, especially the Appendices to LAQM.TG(09).

6.24 In some cases, it may be appropriate to carry out a period of air quality monitoring as part of an air quality assessment. This may be particularly helpful where new relevant exposure is proposed in a location with a complex road layout and/or topography, which may be difficult to model, or where there are no data available to verify the model. Monitoring should ideally be carried out for a minimum of six months using a methodology and locations agreed with the local authority. Where monitoring is carried out for less than a year, the results will need to be adjusted to an annual mean equivalent using the methodology described in the Technical Guidance, LAQM.TG(09). This will add to the uncertainty associated with any model verification and adjustment, and this should be recognised.

Describing the impacts

6.25 It is useful for all parties involved in the planning process to use a consistent approach for the description of the impacts. The EIA process requires the magnitude of changes to be set out and taken into account. In many instances there is also a desire to use a consistent descriptive terminology across all environmental impacts within an environmental statement so that, for example, ecological and noise impacts can be described using the same terminology as applied to air quality. The assessment may use its own set of criteria to define magnitude, but the important matter to be concluded is the likely significant effects of the impacts on air quality. There is, therefore, a two stage process to be followed in the assessment:

- a qualitative or quantitative description of the impacts on local air quality arising from the development; and
- a judgement on the overall significance of the effects of any impacts.

6.26 The impacts are usually assessed at selected 'receptors'. It can also be helpful to present the changes in concentrations

across the study area as a whole, using concentration isopleths on a map of the area, as this will help to inform the decision as to whether the effect is significant or not (by describing the geographical extent over which impacts occur and by helping identify the sensitive receptors that might be affected).

6.27 The framework for the assessment of impacts should be capable of application to all types and scales of development. It cannot simply reflect a response to small scale developments, or conversely, to the largest, and should be able to consider point and diffuse sources as well as traffic impacts, and a wide range of pollutants.

6.28 For air quality impacts arising from surrounding sources on new occupants of a development then the impacts are best described in relation to whether or not an air quality objective will be not be met, or is at risk of not being met. An exceedence of the objective value is likely to be considered as being significant, an aspect that is considered further in paragraph 7.12.

6.29 For air quality impacts arising from surrounding sources on new occupants of a development, then the impacts are best described in relation to whether or not an air quality objective will be not be met, or is at risk of not being met An exceedance of the

Table 6.3: Impact descriptors for individual receptors.

Long term average Concentration at receptor in assessment year	% Change in concentration relative to Air Quality Assessment Level (AQAL)			
	1	2-5	6-10	>10
75% or less of AQAL	Negligible	Negligible	Slight	Moderate
76-94% of AQAL	Negligible	Slight	Moderate	Moderate
95-102% of AQAL	Slight	Moderate	Moderate	Substantial
103-109% of AQAL	Moderate	Moderate	Substantial	Substantial
110% or more of AQAL	Moderate	Substantial	Substantial	Substantial

Explanation

1. AQAL = Air Quality Assessment Level, which may be an air quality objective, EU limit or target value, or an Environment Agency 'Environmental Assessment Level (EAL)'.
2. The Table is intended to be used by rounding the change in percentage pollutant concentration to whole numbers, which then makes it clearer which cell the impact falls within. The user is encouraged to treat the numbers with recognition of their likely accuracy and not assume a false level of precision. Changes of 0%, i.e. .less than 0.5% will be described as Negligible..
3. The Table is only designed to be used with annual mean concentrations.
4. Descriptors for individual receptors only; the overall significance is determined using professional judgement (see Chapter 7). For example, a 'moderate' adverse impact at one receptor may not mean that the overall impact has a significant effect. Other factors need to be considered.
5. When defining the concentration as a percentage of the AQAL, use the 'without scheme' concentration where there is a decrease in pollutant concentration and the 'with scheme;' concentration for an increase.
6. The total concentration categories reflect the degree of potential harm by reference to the AQAL value. At exposure less than 75% of this value, i.e. well below, the degree of harm is likely to be small. As the exposure approaches and exceeds the AQAL, the degree of harm increases. This change naturally becomes more important when the result is an exposure that is approximately equal to, or greater than the AQAL.
7. It is unwise to ascribe too much accuracy to incremental changes or background concentrations, and this is especially important when total concentrations are close to the AQAL. For a given year in the future, it is impossible to define the new total concentration without recognising the inherent uncertainty, which is why there is a category that has a range around the AQAL, rather than being exactly equal to it.

▲ **Image: © Roger Barrowcliffe**

objective value is likely to be considered as being significant, an aspect that is considered further in **paragraph 7.12**

6.30 In the case of the impacts of a development on the surrounding area, a practical way of assigning a meaningful description to the degree of an impact is to express the magnitude of incremental change as a proportion of a relevant assessment level and then to examine this change in the context of the new total concentration and its relationship with the assessment criterion. In this document, the term Air Quality Assessment Level or AQAL has been adopted, as it covers all pollutants, i.e. those with and without formal standards. In many cases, the AQAL will be the air quality objective value. (Note that impacts may be adverse or beneficial, depending on whether the change in concentration is positive or negative.)

6.31 One advantage of this approach is that it avoids the need for individual pollutants to have their own tailored method of assessment. Since air quality standards are set on the basis of harm, it is reasonable to assume that the degree of harm is represented by the margin by which the AQAL is exceeded. This concept is not universally true and many pollutants exert an effect on human health at exposures that are below the standard[32]. It does, however, provide a sound and consistent basis for a framework for the assessment of impacts. Where legislative standards do not exist for a particular pollutant, it is common practice to adopt the Environmental Assessment Level from the Environment Agency's list in its H1 guidance note[33], which can be used as the AQAL.

6.32 The suggested framework for describing the impacts on the basis set out above is set out in **Table 6.3**. The term AQAL is used to include air quality objectives or limit values, where these exist.

6.33 It is recommended that $PM_{2.5}$ is used to assess the impact of combustion sources (including road traffic) rather than PM_{10}. The AQAL[33] for $PM_{2.5}$ is much lower on the basis of the air quality objective and EU limit value and this therefore represents a more conservative approach. Most particulate matter from combustion processes occurs in the smaller size fraction. If, however, PM_{10} is assessed, then **Table 6.3** should be applied using the limit value of 40 $\mu g/m^3$ as an annual mean, or alternatively a derived value for the annual mean based on exceeding 50 $\mu g/m^3$ for more than 35 days in a year (the equation in LAQM.TGO9 shows an annual mean of 32 $\mu g/m^3$ equating to 35 days at or above 50 $\mu g/m^3$).

6.34 The number of significant figures to which concentrations should be reported should reflect the accuracy associated with predicted changes and the knowledge of background concentrations. This is ultimately a compromise between reducing the number in recognition of the uncertainty associated with air quality calculations and the need to contribute to the

decision making process by being able to demonstrate a small but widespread change, if one exists. Three significant figures may be appropriate, e.g. 0.403, 4.03, or 40.3 etc. There may be occasions, however, when it is better to present results to two significant figures, depending on professional judgement regarding the accuracy of the data[34]. Any rounding of the data should only be applied after any calculations have been completed.

6.35 For most road transport related emissions, and diffuse emissions associated with development, long term average concentrations are the most useful for evaluating the severity of impacts. For any point source, some consideration must also be given to the impacts resulting from short term, peak concentrations of those pollutants that can affect health through inhalation. The Environment Agency uses a threshold criterion of 10% of the short term AQAL as a screening criterion for the maximum short term impact. This is a reasonable value to take and this guidance also adopts this as a basis for defining an impact that is sufficiently small in magnitude to be regarded as having an insignificant effect. Background concentrations are less important in determining the severity of impact for short term concentrations, not least because the peak concentrations attributable to the source and the background are not additive.

6.36 Short term concentrations in this context are those averaged over periods of an hour or less. These are exposures that would be regarded as acute and will occur when a plume from an elevated source affects airborne concentrations experienced by a receptor over an hour or less.

6.37 Impacts expressed using an averaging time of a day are not amenable to this form of assessment, since the plume spread will be much too wide over the course of a day, leading to a different kind of exposure to the peak short term concentrations. This is a problem chiefly with respect to PM_{10} and the expression of its limit value and objective. It is preferred that the annual mean AQAL is used for this pollutant.

6.38 Where such peak short term concentrations from an elevated source are in the range 10-20% of the relevant AQAL, then their *magnitude* can be described as small, those in the range 20-50% medium and those above 50% as large. These are the maximum concentrations experienced in any year and the *severity* of this impact can be described as slight, moderate and substantial respectively, without the need to reference background or baseline concentrations. That is not to say that background concentrations are unimportant,but they will, on an annual average basis, be a much smaller quantity than the peak concentration caused by a substantial plume and it is the contribution that is used as a measure of the impact, not the overall concentration at a receptor. This approach is intended to be a streamlined and pragmatic assessment procedure that avoids undue complexity.

6.39 In most cases, the assessment of impact severity for a proposed development will be governed by the long-term exposure experienced by receptors and it will not be a necessity to define the significance of effects by reference to short-term impacts. The severity of the impact will be substantial when there is a risk that the relevant AQAL for short-term concentrations is approached through the presence of the new source, taking into account the contribution of other prominent local sources.

[28] Integrated Pollution Prevention and Control.

[29] Taking account of criteria published in: a) The Town and Country Planning (Development Management Procedure) (England) Order 2010 – 2010 No. 2184 [(Wales) Order 2012, No 801(W11)] (HMSO), b) The GLA's Supplementary Planning Guidance (SPG) on Sustainable Design and Construction (2014) and c) The Sussex Air Quality Partnership's Air Quality and Emissions Mitigation Guidance for Sussex Authorities (2013) v January 2014. The latter still requires a calculation of emissions even if an assessment is not required.

[30] See www.defra.gov.uk/environment/airquality/local/guidance/pdf/tech-guidance-laqm-tg-09.pdf.

[31] http://iaqm.co.uk/text/guidance/construction-dust-2014.pdf.

[32] This is in part reflected in the description of impacts as being 'slight' or 'moderate', even when concentrations are well below the AQAL (see **Table 6.3**).

[33] An annual mean of 20 μg/m³ for $PM_{2.5}$, by 1 January 2020, compared with 40 μg/m³ for PM_{10}.

[34] This is not the case where the changes being reported are small, as it could lead to the presentation of misleading data. For example a change of 0.2 μg/m³ for the annual mean nitrogen dioxide concentration from 40.2 to 40.4 μg/m³ would be presented as 40 μg/m³ without and 40 μg/m³ with the scheme, while the same change applied to 40.4 μg/m³ without to 40.6 μg/m³ with the scheme, would be presented as 40 μg/m³ without and 41 μg/m³ with the scheme.

7. Assessing Significance

7.1 Impacts on air quality, whether adverse or beneficial, will have an effect on human health that can be judged as 'significant' or 'not significant'. This is the primary requirement of the EIA regulations, but is also relevant to other air quality assessments. It is important to distinguish between the meaning of 'impact' and 'effect' in this context. An impact is the change in the concentration of an air pollutant, as experienced by a receptor. This may have an effect on the health of a human receptor, depending on the severity of the impact and other factors that may need to be taken into account. Judging the severity of an impact is generally easier than judging the significance of an effect.

7.2 The significance of effect that any proposed development might have will also be judged at two separate stages of the development control process, as follows:

- the first is within the air quality report accompanying the planning application; while
- the second is when the local authority's air quality specialist makes his/her recommendations to the planning officer.

7.3 These are mutually exclusive requirements serving different purposes. Ultimately, any disputes on these matters are dealt with by the judgement of the planning committee and/or a planning inspector following a planning appeal. A significant air quality effect is not, of itself, a reason for refusal of a planning application; that decision will be the outcome of a careful consideration of a number of factors by a planning committee (or a planning inspector/Secretary of State), air quality being just one of the factors.

7.4 The assessment framework for describing impacts can be used as a starting point to make a judgement on significance of effect, but there will be other influences that might need to be accounted for. The impact descriptors set out in **Table 6.3** are not, of themselves, a clear and unambiguous guide to reaching a conclusion on significance. These impact descriptors are intended for application at a series of individual receptors. Whilst it may be that there are 'slight', 'moderate' or 'substantial' impacts at one or more receptors, the overall effect may not necessarily be judged as being significant in some circumstances.

7.5 One of the relevant factors in the judgement of the overall significance of effect may relate to the potential for cumulative impacts and, in such circumstances, several impacts that are described as 'slight' individually could, taken together, be regarded as having a significant effect for the purposes of air quality management in an area, especially where it is proving difficult to reduce concentrations of a pollutant. Conversely, a 'moderate' or 'substantial' impact may not have a significant effect if it is confined to a very small area and where it is not obviously the cause of harm to human health.

7.6 Often, it is possible to be very clear when an impact is sufficiently slight that it has no effect on receptors and can therefore be described unequivocally as 'not significant'[35]. In the opposite case, when an impact is clearly substantial, it will be obvious that there is potential for a significant effect. The problem lies in the intermediate region where there is likely to be uncertainty on the transition from insignificant to significant. In those circumstances where a single development can be judged in isolation, it is likely that a 'moderate' or 'substantial' impact will give rise to a significant effect and a 'negligible' or 'slight' impact will not have a significant effect, but such judgements are always more likely to be valid at the two extremes of impact severity.

7.7 Any judgement on the overall significance of effect of a development will need to take into account such factors as:

- the existing and future air quality in the absence of the development;
- the extent of current and future population exposure to the impacts; and
- the influence and validity of any assumptions adopted when undertaking the prediction of impacts.

Other factors may be relevant in individual cases.

7.8 A judgement of the significance should be made by a competent professional who is suitably qualified. The reasons for reaching the conclusions should be transparent and set out logically. Whilst the starting point for the assessment of significance is the degree of impact, as defined by **Table 6.3**, this should be seen as one of the factors for consideration, not least because the outcome of this assessment procedure applies to a receptor and not the overall impact.

7.9 The judgement on significance relates to the consequences of the impacts; will they have an effect on human health that could be considered as significant? In the majority of cases, the impacts from an individual development will be insufficiently large to result in measurable changes in health outcomes that could be regarded as significant by health care professionals. In reality, therefore, it is the impact on local air quality that is ©used as a proxy for assessing effects on health.

7.10 There are no viable means of assigning significance to health outcomes as yet, even though quantification of mortality and morbidity effects from certain air pollutants at the population

▲ **Image: © Roger Barrowcliffe**

level has evolved to a point where reasonable confidence in the estimates exist. Part of any judgement on the significance of health effects would incorporate the size or the population exposed to changes in concentrations. Any judgement on the significance of effects on health is part of a Health Impact Assessment and not the air quality assessment being described here.

7.11 For local authorities, there may also be a question of meeting air quality objectives as part of their obligations under Local Air Quality Management Regulations. As has already been noted, the presence of an AQMA that may be affected by a proposed development will increase the sensitivity of the application and any accompanying assessment. The impacts descriptor table acknowledges this and points to a conclusion of significant effect in cases where concentrations of a regulated pollutant are in excess of the objective value. Where the baseline concentrations are close to the objective value at a receptor, but not exceeding it, a case may be made for the development's predicted contribution being significant. It will always be difficult, however, to attribute the exceedance of an objective to any individual source.

7.12 The effect on the residents or occupants of any new development where the air quality such that an air quality objective at the building façade is not met will be judged as significant, unless provision is met to reduce their exposure by some means. For people working at new developments in this situation, the same will not be true as occupational exposure standards are different, although any assessment may wish to draw attention to the undesirability of the exposure.

[35] The precise role of the development control process in delivering compliance with the EU limit values is uncertain, and clarification has been sought from Defra. Until unambiguous clarification from a Government department is provided on this matter, which confirms that any increase in concentration should not be permitted where an EU limit value is not met, then the precise extent to which an impact may need to be regarded as a significant effect in this context and in such areas is unclear.

Abbreviations and acronyms

AADT: Annual Average Daily Traffic

AADT: Annual average daily traffic

AQA: Air quality assessment

AQAL: Air Quality Assessment Level

AQAP: Air Quality Action Plan

AQMA: Air Quality Management Area

AQO: Air Quality Objective

CHP: Combined Heat and Power (Plant)

Defra: Department for Environment, Food and Rural Affairs

EA: Environment Agency

EIA: Environmental Impact Assessment

EPR: Environmental Permitting (England and Wales) Regulations 2010

EPUK: Environmental Protection UK

EU: European Union

EV: Electric vehicle

HDV: Heavy Duty Vehicle

HRAPIE Health risks of air pollution in Europe – A World Health Organisation project

IAQM: Institute of Air Quality Management

LA: Local Authority

LAQM: Local Air Quality Management

LAQM.TG(09): Local Air Quality Management: Technical Guidance (09)

LDF: Local Development Frameworks

LDP: Local Development Plans

LDV: Light duty vehicle

LPA: Local planning authority

NO_2: Nitrogen dioxide

NO_x: Oxides of nitrogen

NPPF: National Planning Policy Framework

NRW: Natural Resources Wales

PM_{10} and $PM_{2.5}$: Particulate matter with an aerodynamic diameter of less than 10 microns (m) (PM_{10}) or less than 2.5µm ($PM_{2.5}$), expressed in units of µg/m^3.

PPG: Planning Practice Guidance

PPW: Planning Policy Wales

PHE: Public Health England

SPD: Supplementary Planning Document.

SPG: Supplementary Planning Guidance.

SO_2: Sulphur dioxide

TAN: Technical Advice Note

VPD: Vehicles per day

WG: Welsh Government

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The Impact on Health of Emissions to Air from Municipal Waste Incinerators

September 2009

Summary

The Health Protection Agency has reviewed research undertaken to examine the suggested links between emissions from municipal waste incinerators and effects on health. While it is not possible to rule out adverse health effects from modern, well regulated municipal waste incinerators with complete certainty, any potential damage to the health of those living close-by is likely to be very small, if detectable. This view is based on detailed assessments of the effects of air pollutants on health and on the fact that modern and well managed municipal waste incinerators make only a very small contribution to local concentrations of air pollutants. The Committee on Carcinogenicity of Chemicals in Food, Consumer Products and the Environment has reviewed recent data and has concluded that there is no need to change its previous advice, namely that any potential risk of cancer due to residency near to municipal waste incinerators is exceedingly low and probably not measurable by the most modern techniques. Since any possible health effects are likely to be very small, if detectable, studies of public health around modern, well managed municipal waste incinerators are not recommended.

The Agency's role is to provide expert advice on public health matters to Government, stakeholders and the public. The regulation of municipal waste incinerators is the responsibility of the Environment Agency.

Introduction

1. The use of incineration for waste disposal in the UK is increasing. Applications for permits to build and operate incinerators give rise to local concerns about possible effects on health of emissions. Responsibility for the environmental permitting of municipal waste incinerators lies with the Environment Agency. The Health Protection Agency (HPA) has a statutory responsibility to advise Government and Local Authorities on possible health impacts of air pollutants.

2. The operators of modern waste incinerators are required to monitor emissions to ensure that they comply, as a minimum, with the limits in the EU Waste Incineration Directive (2000/76/EC), which sets strict emission limits for pollutants. This Directive has been implemented in England and Wales by the Environmental Permitting (EP) (England and Wales) Regulations 2007 (note

that from April 2008 these replaced the Pollution Prevention and Control (PPC) (England and Wales) Regulations 2000).

3. Under the EP Regulations, the operator is required to apply for an environmental permit. Consideration of this application will include such issues as health effects and organisations such as the local Primary Care Trust (PCT); the HPA and Food Standards Agency (FSA) are usually consulted. The permit itself will set out strict operating requirements which must be complied with, this will include monitoring. Should a breach of the permit occur, action may be taken by the regulator.

4. Applications to build and operate incinerators invariably include an assessment of likely emissions to air. Modern incinerators emit only small amounts of chemicals to air (see para 16) in comparison with older incinerators and, although no absolute assurance of a zero effect on public health can be provided, the additional burden on the health of the local population is likely to be very small. Studies published in the scientific literature showing health effects in populations living around incinerators have, in general, been conducted around older incinerators with less stringent emission standards and cannot be directly extrapolated with any reliability to modern incinerators (see paras 6 and 26)

5. The incineration process can result in three potential sources of exposure, (1) emissions to the atmosphere, (2) via solid ash residues, and (3) via cooling water. Provided that solid ash residues and cooling water are handled and disposed of appropriately, atmospheric emissions remain the only significant route of exposure to people. This paper is thus concerned only with the health effects of emissions to air.

6. The comparative impacts on health of different methods of waste disposal have been considered in detail in a report prepared for the Department of Environment, Food and Rural Affairs (Defra 2004). This work was undertaken by a group of consultants led by the independent consultants Enviros and included experts in the air pollution field. The report was reviewed by The Royal Society and its comments were incorporated by the authors of the report. This report is the most extensive available in the field and concludes that well managed, modern incinerators are likely to have only a very small effect on health. Since the evidence base has not changed significantly since 2004 it would be an inefficient use of resources to repeat the work undertaken by Enviros (see above) for Defra when applications to build and operate individual incinerators are being considered. The HPA's view is that the study undertaken for Defra by Enviros can be relied on although, like all scientific findings, it may be subject to revision if new data were to emerge.

7. Concerns about possible effects on health of emissions to air tend to focus on a few well known pollutants: particles, polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzo-*p*-furans (commonly referred to as "dioxins") and other carcinogens such as the polycyclic aromatic hydrocarbons (PAH). Much is known about the effects on health of these

compounds. Detailed reports prepared by expert advisory committees are available: these include reports by the Department of Health's Committee on the Medical Effects of Air Pollutants (COMEAP) on particulate matter (COMEAP, 1995, 1998, 2001a, 2009); by Defra's Expert Panel on Air Quality Standards (EPAQS) on benzene, 1,3-butadiene (reports 1 and 2), particles (reports 1 and 2), PAH compounds, and metals and metalloids[1] (Department of the Environment, 1994a,b, 1995; Department of the Environment Transport and the Regions, 1999, 2001; Department for the Environment, Food and Rural Affairs, 2002, 2009) and the Committee on the Toxicity of Chemicals in Food, Consumer Products and the Environment's statement on dioxins and dioxin-like polychlorinated biphenyls (Committee on Toxicity, 2001).

Particles

8. Questions are often asked about the possible effects on health of particles emitted by incinerators. The Committee on the Medical Effects of Air Pollutants (COMEAP) has published a series of statements and reports on the effects of air pollutants on health in the UK. It is accepted that exposure to current levels of common air pollutants damages health. The Air Quality Strategy for England, Scotland, Wales and Northern Ireland seeks to reduce concentrations of air pollutants. Where concentrations of air pollutants are raised, Air Quality Management Areas are defined and plans to reduce concentrations are developed by Local Authorities. Details of the Air Quality Strategy can be found on the Defra website: http://www.defra.gov.uk/environment/airquality/strategy/index.htm

9. Both long-term exposure and short-term increases in exposure to particles can damage health. This is widely accepted (World Health Organization, 2006). Long term exposure affects the risk of mortality, especially from cardiovascular disease and from lung cancer (COMEAP, 2009, COMEAP, 2006; Health Effects Institute, 2000). Short-term increases in concentrations cause cardio-respiratory effects including an increase in deaths from heart attacks and from respiratory disease, increased hospital admissions for treatment of these disorders and increases in related symptoms. No thresholds of effect can be identified for either the effects of long-term exposure or for the effects of short-term increases in concentrations. Thus, any increase in particle concentrations should be assumed to be associated with some effect on health. The critical step in assessment of effects on health is not simply making the correct assertion that some effect is possible but in estimating the size of that effect. This is discussed below.

10. Evidence of the effects of particles on health comes, in the main, from epidemiological studies. For the effects of long-term exposure attention has been focused on $PM_{2.5}$; for the effects of short-term increases in concentrations both $PM_{2.5}$ and PM_{10} have been extensively used as metrics of the ambient aerosol. PM_{10} is defined as the mass of particles of less than

[1] Arsenic, chromium, nickel and beryllium

(about) 10 microns in diameter per cubic metre of air. $PM_{2.5}$ is an analogous measure: in this case, the mass of particles of less than about 2.5 microns in diameter per cubic metre of air. The exact definitions are given in the recent Defra report on ambient particles (Defra, 2005). The exact mechanisms of effect of particles on health are incompletely understood but several plausible hypotheses are being pursued; the generation of free radicals in the respiratory system and more widely in the body, the induction of an inflammatory response in the lung, effects on clotting factors in the blood, effects on the rate of development of atherosclerotic plaques in coronary arteries and effects on the regulation of the heart beat are all being studied intensively. It is possible that metals found in association with particles play an important role. It is also possible that the ultrafine component of the ambient aerosol plays an important role. These, and other, possibilities are not yet proven.

11. The lack of a complete understanding of the mechanisms of effects of particles does not prevent prediction of the effects on health of increased concentrations of particles monitored as PM_{10} and/or $PM_{2.5}$. Meta-analytical techniques have been applied to the results of primary studies and summary coefficients linking PM_{10} and $PM_{2.5}$ with effects on health have been derived (COMEAP, 1998, 2009; World Health Organization, 2006). If these coefficients are applied to the small increases in concentrations of particles produced, locally, by incinerators, the estimated effects on health are likely to be small. This is because the coefficients themselves are small, the increase in concentration due to operation of the incinerator is likely to be small, and so is the size of the potentially exposed local population.

12. It is sometimes claimed that the “wrong particles” are considered when estimating the possible effects on health of emissions from incinerators. It should be understood that impact calculations of the effects on health of emissions from incinerators are done by using the coefficients derived from epidemiological studies. Because we do not know with certainty the active components of the ambient aerosol, coefficients linking effects on health with changes in mass concentrations (PM_{10} and/or $PM_{2.5}$) are used in the impact calculations. At present we have no clear epidemiological evidence to distinguish between the toxicity of samples of particles collected for PM_{10} or $PM_{2.5}$ measurements in different areas. National policy (Defra, 2007a,b) and the EC Directive on Ambient Air Quality and Cleaner Air for Europe (European Parliament and Council of the European Union, 2008) are based on the assumption that particles collected for PM_{10} and $PM_{2.5}$ measurements do not differ in their effects on health from place to place. In this context it is worth noting that PM_{10} and $PM_{2.5}$ samples from around the world can vary substantially in their chemical composition and size distribution but nonetheless exhibit similar concentration-response coefficients in time-series epidemiological studies. It is accepted that this view could change and that monitoring of chemical characteristics of the ambient aerosol (for example, its metallic components), the number of particles per unit of volume of air, the total surface area of particles per unit volume of air, or the capacity of particles to generate free radicals could prove more valuable than measurements of mass concentrations (PM_{10} and $PM_{2.5}$). But none of this is yet well

established and international and national regulations are currently framed in terms of mass concentrations. It seems reasonable that these regulations and the approaches upon which they are based should be applied to considerations of the effects on health of particles emitted by incinerators. It may be asked why studies of the specific impacts on health of the small increases in local concentrations of particles produced by incinerators are not done routinely. The main reason for this is that the concentration increment produced by incinerators is likely to be too small to allow an impact on health to be identified in the local population.

13. It is sometimes claimed that PM_{10} measurements ignore particles most likely to be deposited in the lung, or, more specifically, in the gas exchange zone of the lungs. This is incorrect and stems from a misunderstanding of the term PM_{10}. Tapered element oscillating microbalance (TEOM) monitors are equipped with a sampling head that selects essentially all particles of less than 10 μm aerodynamic diameter. PM_{10} measurement is designed to collect effectively all those particles small enough to pass the upper airways (nose, mouth, pharynx, larynx) and thus of a size that allows a chance of deposition in the lung. $PM_{2.5}$ is intended to represent that fraction of the aerosol with a high probability of deposition in the gas exchange zone of the lung in vulnerable individuals. It will be obvious that PM_{10} includes $PM_{2.5}$ and that $PM_{2.5}$ cannot exceed PM_{10} in any given sample of air.

14. It is sometimes, further, claimed that PM_{10} or $PM_{2.5}$ do not include nanoparticles present in the air. This is also incorrect. Nanoparticles are efficiently collected by PM_{10} and $PM_{2.5}$ samplers but make only a small contribution to the results expressed as PM_{10} or $PM_{2.5}$. If particles of less than 100 nm diameter alone were collected from a known volume of air and weighed, the resulting concentration could be expressed as $PM_{0.1}$ (100 nm = 0.1 microns). In a sample of air collected in a UK urban area on a typical day we might expect results similar to those given below:

PM_{10}	20 μg/m^3
$PM_{2.5}$	13 μg/m^3
$PM_{0.1}$	1-2 μg/m^3

PM_{10} includes and exceeds $PM_{2.5}$ which in turn includes and exceeds $PM_{0.1}$.

15. It is quite correct to say that nanoparticles make a large contribution to the number of particles per unit volume of air. Particles of less than about 500 nm in diameter dominate the number concentration of ambient particles. It might be correctly suggested that if a specified source, for example an incinerator, produced mainly nanoparticles, changes in local mass concentrations (PM_{10} and to a lesser extent $PM_{2.5}$) would not reflect the increase in numbers of particles in the air. We do not, however, know how to interpret measurement of number concentrations of particles in health terms. Work in this area is developing. It may be that, although the evidence is as yet weak in comparison with that relating to mass concentrations, particle numbers will link with some effects on health better than mass concentrations. No generally accepted coefficients that allow the use of number

concentrations in impact calculations have yet been defined. As stated above, regulations are currently framed in terms of mass concentrations and it is unreasonable to expect local health professionals to interpret number concentrations in quantitative health terms when national experts have not yet judged that the evidence is sufficient to do so. COMEAP will be looking at whether quantification of the effects of particle number concentrations is possible as part of its work on the quantification of the health effects of air pollution. No Air Quality Standards are defined in terms of number concentrations of particles.

16. The contribution made by waste incineration to national emissions of particles is low. Data provided by Defra (National Emissions Inventory www.naei.org.uk) show that 2006 national emissions of PM_{10} from waste incineration are 0.03% of the total compared with 27% and 25% for traffic and industry respectively[2]. This low proportion is also found at a local level – the Environment Agency have informed HPA of one incinerator modelling study that found a modelled ground level increment in PM_{10} of 0.0005 $\mu g/m^3$ as an annual average (Environment Agency, 2009). The increment in $PM_{2.5}$ could not exceed this, and would be likely to be lower. In addition, Defra is expanding its general $PM_{2.5}$ monitoring and will scrutinise this to see if any individual sources make a noticeable addition to measured concentrations.

17. Questions are often asked about the effects of air pollutants, including those emitted by waste incineration, on children's health. The World Health Organization (WHO) in its 2005 report on Air Pollution and Children's Health and Development, concluded that there was an association between air pollution and infant mortality that appeared to be mainly due to particulate air pollution. COMEAP, in a 2008 statement on Air Pollution and Children's Health, endorsed WHO's general conclusions although the COMEAP statement does not comment on which pollutant is likely to be responsible. Annexes to the statement indicate that, of the studies published since the WHO report, some find effects of particulate air pollution and some do not. Metrics of particulate air pollution used in these studies included PM_{10} and total suspended particulates, as well as $PM_{2.5}$. The size of the effects reported in these studies relates to large changes in $PM_{2.5}$, larger than would be expected to be caused by the operation of an incinerator. Given the small effects of incinerators on local concentrations of particles, it is highly unlikely that there will be a detectable effect of any particular incinerator on local infant mortality.

18. When carrying out studies which investigate health effects around point sources of pollution such as incinerators, or when mapping health effects around such sources, it is important to control for other factors which can influence the health outcomes under investigation before drawing any conclusions. So when investigating the effect of a source of $PM_{2.5}$ emissions on infant mortality rates, it would be important to control for other sources of $PM_{2.5}$ emissions, and for factors which are known to influence infant mortality

[2] National Atmospheric Emissions Inventory PM_{10} http://www.naei.org.uk/emissions/emissions_2006/summary_tables.php?action=unece&page_name=PM1006.html'

rates, for example, socio-economic factors or ethnicity. Maps showing death rates or levels of morbidity are useful in raising hypotheses, but they do not supply evidence of cause and effect.

Carcinogens

19. Chemicals which cause cancer are described as carcinogens. For risk assessment purposes, carcinogens are divided into two groups depending on their mechanism of action:

(a) Genotoxic carcinogens: these induce cancer by a mechanism that involves the compound itself, or a metabolite, reacting directly with the genetic material of cells (DNA), producing a mutation. This process is called mutagenicity. It is theoretically possible that one "hit" on DNA may produce a mutation that can eventually develop into a tumour. The assumption is thus made for genotoxic carcinogens that they do not have a threshold and that any exposure is associated with an increase in risk, albeit this may be very small. Most of the known human chemical carcinogens are in this group, e.g. aflatoxins, benzene, 1,3-butadiene, 2-naphthylamine, polycyclic aromatic hydrocarbon (PAH) compounds.

(b) Non-genotoxic carcinogens: these induce cancer by mechanisms that are not based on mutagenicity. These chemicals give negative results in the well recognised tests for mutagenicity. Unlike the genotoxic carcinogens, which are characterised by a common mechanism, there are a number of different mechanisms involved. Examples include sustained cell proliferation in a sensitive tissue (resulting in expression of a spontaneous mutation) due to cytotoxic effects, hormonal stimulation or immunosuppression. These effects have a threshold based on the precursor toxicological effect such as cytotoxicity, i.e. there is a level of exposure below which they do not have an effect. Examples of such compounds are oestrogens and 2,3,7,8-tetrachlorodibenzo-para-dioxin (TCDD or "dioxin").

20. In the air pollution field, genotoxic carcinogens are the major focus of interest. In the following discussion, the term "carcinogens" is used to represent genotoxic carcinogens.

21. The carcinogenic effects of PAH compounds can be identified by means of studies in experimental animals only at very much higher concentrations than occur in ambient air. These high exposures are necessary because practical limitations regarding the number of animals used in these tests mean that they cannot reliably detect increases in tumour incidence below a few percent. However, for public health purposes, the principal concern is about effects that occur at a much lower incidence in the human population, but are undetectable in animal studies. The calculation of cancer risk at low environmental exposures from mathematical modelling of

the results from the high dose animal data presents great difficulty. The expert advisory committee, the Committee on the Carcinogenicity of Chemicals in Food, Consumer Products and the Environment (COC) has consistently expressed concern at the use of such modelling to extrapolate to levels of exposure that are orders of magnitude lower than the observed range. This was most recently stated in the 2004 guidelines. (The reasons are based on the fact that the various models available do not take into account the biological complexity of the carcinogenesis process, the extrapolations are based on a few data points over a very narrow and high dose range, and very wide variations in risk estimates are produced depending on the models used. Their use gives an impression of precision that cannot be justified). The COC does not recommend their use for routine risk assessment.

22. In some cases, carcinogenic effects have been demonstrated in epidemiological studies in humans. Such studies have almost always involved occupational exposure where workplace levels in the past may have been much higher than those in ambient air. It is difficult to demonstrate the effects of exposure to ambient concentrations of carcinogens (the concentrations are so low that vast numbers of people would need to be studied to produce clear results) but such effects are assumed to be possible, on the grounds that there is no threshold for the effects of many of these compounds. If good quality epidemiological studies are available it is possible to derive models of the relationship between exposure and effect that allow prediction, with some confidence, of likely cancer incidence at ambient concentrations. It should be noted, however, that the actual accuracy of such predictions cannot be assessed and such extrapolations still involve some considerable uncertainty and should be used with caution.

23. The Expert Panel on Air Quality Standards (EPAQS) has recommended air quality standards for benzene, 1,3-butadiene and PAH compounds using a different approach from that used by the World Health Organization (WHO), which is based on quantitative risk assessment. This is because of the concerns of the COC regarding the use of mathematical models to estimate cancer risk. Indeed, the COC endorsed the approach used by EPAQS. This involved the application of Uncertainty Factors to the results of studies of the effects on man of exposure to high concentrations of the carcinogens specified above. Standards derived in this way do not offer a complete guarantee of safety (this is impossible with non-threshold compounds) but do define concentrations at which the risks to health are likely to be very small and unlikely to be detectable. If it is found that incinerators emit the carcinogens considered by EPAQS, it is reasonable to compare the augmented local concentration (i.e. the local background concentration plus the increment contributed by the incinerator) with the EPAQS standard. If this is not exceeded it may be reasonably assumed that the additional risk imposed by the emissions is minimal. If, on the other hand, the emissions cause the local concentrations to exceed the EPAQS standard(s), the appropriate regulator would need to decide whether the additional risk posed by the incinerator was a cause for concern and what further reductions may be necessary.

Dioxins

24. It is recognised that there are particular concerns about emissions of dioxins from incinerators. The HPA and DH are advised on the health effects of such compounds by the independent expert advisory committee, the Committee on the Toxicity of Chemicals in Food, Consumer Products and the Environment (COT). The COT has recommended a tolerable daily intake (TDI) for dioxins, which is the amount which can be ingested daily over a lifetime without appreciable health risk. This TDI is based on a detailed consideration of the extensive toxicity data on the most well studied dioxin, TCDD, but may be used to assess the toxicity of mixtures of dioxins and dioxin-like PCBs by use of Toxic Equivalency Factors, which allow concentrations of the less toxic compounds to be expressed as an overall equivalent concentration of TCDD. These toxicity-weighted concentrations are then summed to give a single concentration expressed as a Toxic Equivalent (TEQ). The system of Toxic Equivalency Factors (TEFs) used in the UK and a number of other countries is that set by the World Health Organization (WHO)[3], and the resulting overall concentrations are referred to as WHO-TEQs (van den Berg, 2006). Thus, the COT has recommended a tolerable daily intake for dioxins of 2 picograms WHO-TEQ/kg body weight/day based on the most sensitive effect of TCDD in laboratory animals, namely, adverse effects on the developing fetus resulting from exposure *in utero*. As this was the most sensitive effect it will protect against the risks of other adverse effects including carcinogenicity. The advice of the other sister committees, COC and the Committee on Mutagenicity of Chemicals in Food, Consumer Products and the Environment (COM), informed the conclusion, namely that dioxins do not directly damage genetic material and that evidence on biological mechanisms suggested that a threshold based risk assessment was appropriate. The full statement is available (COT, 2001).

25. The majority (more than 90%) of non-occupational human exposure to dioxins occurs via the diet, with animal-based foodstuffs like meat, fish, eggs, and dairy products being particularly important. Limited exposure may also occur via inhalation of air or ingestion of soil depending on circumstances. Regarding emissions from municipal waste incinerators, the current limit for dioxins and furans is 0.1 nanogram per cubic metre of emitted gases. A nanogram is one thousand millionth of a gram. Inhalation is a minor route of exposure and, given that Defra has calculated that incineration of municipal solid waste accounts for less than 1% of UK emissions of dioxins[4], the contribution of incinerator emissions to direct respiratory exposure of dioxins is a negligible component of the average human intake. However, dioxins may make a larger contribution to human exposure via the food chain, particularly fatty foods. Dioxins from emissions could also be deposited on soil and crops and accumulate in the food chain via animals that graze on the pastures,

[3] Note: The Waste Incineration Directive (2000/76/EC) sets Air Emission Limit Values for dioxins using a slightly different system of TEQs i.e. international- or I-TEQs, which vary slightly from WHO-TEQs.

[4] Review of Environmental and Health Effects of Waste Management: Municipal Solid Waste and Similar Wastes**.** Extended Summary. Enviros, University of Birmingham and Defra. May 2004.

though dioxins are not generally taken up by plants. Thus the impact of emissions on locally produced foods such as milk and eggs is considered in deciding whether to grant a permit. These calculations show that, even for people consuming a significant proportion of locally produced foodstuffs, the contribution of incinerator emissions to their intake of dioxins is small and well below the tolerable daily intake (TDI) for dioxins recommended by the relevant expert advisory committee, Committee on Toxicity of Chemicals in Food, Consumer (see http://cot.food.gov.uk/cotstatements/cotstatementsyrs/cotstatements2001/dioxinsstate).

Epidemiological studies: municipal waste incinerators and cancer

26. The COC has issued two statements on the cancer epidemiology of municipal waste incinerators. The initial statement followed a review of a large study by the Small Area Health Statistics Unit which examined cancer incidence between the mid 1970s and the mid 1980s in 14 million people living within 7.5 km of 72 municipal solid waste incinerators in Great Britain[5] (Elliott *et al*, 1996; COC, 2000). Prior to this there had been very few studies of cancer mortality around municipal waste incinerators and none in the UK. The incinerators studied by Elliott *et al* (1996) were the older generation operating prior to introduction of strict emission controls and were more polluting than modern incinerators. After considering this study, the COC concluded that: "*any potential risk of cancer due to residency (for periods in excess of 10 years) near to municipal solid waste incinerators was exceedingly low, and probably not measurable by the most modern techniques"* (COC, 2000).

27. In 2008, the Committee reviewed seven new studies on cancer incidence near municipal solid waste incinerators which had been published since 2000 (Comba et al, 2003; Floret et al, 2003; Knox E, 2000; Viel et al, 2000; 2008a and 2008b; Zambon et al, 2007). All had studied the older generation of incinerator and three studies were of an incinerator for which emissions of dioxins were reported to have exceeded even the older emission standard. There were problems interpreting most of these studies due to factors such as failure to control for socio-economic confounding or inclusion of emission sources other than municipal waste incinerators. The COC concluded that *"Although the studies indicate some evidence of a positive association between two of the less common cancers i.e. non-Hodgkin's lymphoma and soft tissue sarcoma and residence near to incinerators in the past, the results cannot be extrapolated to current incinerators, which emit lower amounts of pollutants. …Moreover, they are inconsistent with the results of the larger study…carried out by the Small Area Health Statistics Unit."* It concluded that there was no need to change its previous advice but that the situation should be kept under review (COC, 2009).

[5] These included all known municipal incinerators which opened before 1976. Incinerators starting from 1976 were excluded, to ensure an appropriate lag period for development of any cancer associated with the emissions.

Conclusions

28. Modern, well managed incinerators make only a small contribution to local concentrations of air pollutants. It is possible that such small additions could have an impact on health but such effects, if they exist, are likely to be very small and not detectable. The Agency, not least through its role in advising Primary Care Trusts and Local Health Boards, will continue to work with regulators to ensure that incinerators do not contribute significantly to ill-health.

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Glossary

Aflatoxins
Naturally occurring toxins produced by the fungus *Aspergillus sp.*

Aerodynamic diameter
The actual diameter of a spherical particle of unit density with the same terminal velocity as the particle under consideration. The term aerodynamic diameter allows particles of differing densities and shapes to be compared in terms of their likelihood of depositing in the lung.

Air Quality Standard (AQS)
The concentration of a pollutant (expressed, generally, as mass per unit volume) and qualified by an averaging time, regarded as acceptable by an Expert Group or other standard setting body. Air Quality Standards do not provide an absolute guarantee of safety for health.

Ambient aerosol
An aerosol is a suspension of fine particles or liquid droplets in a gas. Ambient refers to the surroundings. In the air pollution context, this refers to the suspension of fine particles in the general outdoor air.

Atherosclerotic plaques
The discrete lesions of the arterial wall in atherosclerosis i.e., disease of the blood vessels involving the accumulation of fatty material in the inner layer of the arterial wall resulting in narrowing of the artery. These fatty deposits are known as plaques.

1,3-butadiene
An industrial chemical used in the production of synthetic rubber. It is also produced by the combustion of petrol and diesel. It is efficiently removed by catalytic convertors.

Carcinogens
Agents that cause cancer. Chemical carcinogens are chemicals that may produce cancer.

Cell proliferation
An increase in the number of cells as a result of cell growth and cell division.

Clotting factors
Substances (proteins) in blood that act in a complex series of reactions to stop bleeding by forming a clot.

Coefficients
A constant multiplication factor. For example, a health effect might increase by 0.5% for every unit increase in the concentration of a pollutant. This can be derived as the slope from a graph relating health effects and pollutant concentrations.

Coronary arteries
The network of blood vessels that supply heart muscle with oxygen-rich blood.

Cytotoxic
Toxic to cells.

Dioxins
This refers to a large group of chemicals with similar chemical structure (chlorinated dibenzo-p-dioxins and chlorinated dibenzo-p-furans). They vary greatly in toxicity, some being very toxic, others showing a similar pattern of toxicity but of lower potency. They are not produced commercially but are formed in small amounts in most forms of combustion (fires etc.). The most studied compound in this series is the highly toxic TCDD (2,3,7,8-tetrachlorodibenzo-p-dioxin).

Dioxin-like PCBs
Polychlorinated Biphenyls (PCBs) are another group of substances, some of which have similar biological activity to dioxins. These are referred to as Dioxin-like PCBs. There are many other PCBs that do not have dioxin-like properties.

Epidemiological studies
Studies of the distribution and the aetiology (causes) of disease in humans.

Free radicals
Highly reactive chemical structures (due to the presence of a chemical species that has lost an electron and thus contains an unpaired electron in the outer shell of the molecule). They are unstable and can react in biological systems with nearby substances such as lipids, proteins or DNA producing damage.

Furans
Chemicals related to furan. Furan contains carbon, hydrogen and oxygen with the carbon atoms and an oxygen atom forming a 5 sided ring.

Gas exchange zone
The part of the lung in which oxygen diffuses from the air to the blood and carbon dioxide diffuses from the blood to the air. The alveoli, alveolar ducts and respiratory bronchioles make up the gas exchange zone.

Immunosuppression
Suppression of the immune system.

Incidence
New occurrence of a disease over a specified time period.

In-utero
In the uterus (womb).

Larynx
Dilated region of the airway above the upper end of the trachea or windpipe. The vocal cords lie within the larynx.

Mass concentration of particles
The mass of particles per unit volume of air. Usually expressed as $\mu g/m^3$ (micrograms per cubic metre).

Metabolite
Chemicals that enter the body can be changed by processes in the body into different chemicals. These are described as metabolites of the original chemical.

Metalloid
An element that is not clearly a metal or non-metal but has some intermediate properties in terms of malleability, ductility, conductivity and lustre. The following elements are generally considered to be metalloids: boron; silicon; germanium; arsenic; antimony; tellurium; polonium.

Meta-analysis
In the context of epidemiology, a statistical analysis of the results from independent studies which aims to produce a single estimate of an effect.

Metric
A measure for something. PM_{10} is a measure (or metric) of the concentration of particles in the air.

Microgram (µg)
One microgram is 1×10^{-6}g. There are 1,000,000 (1 million) micrograms in a gram.

Micron (µm)
This is a unit of length that equals one thousandth of a millimetre.

Mortality
Deaths.

Mortality rate
The number of deaths in a population.

Morbidity
Ill health.

Mutation
A permanent change in the amount or structure of the genetic material (DNA) in a cell or organism which can result in a change in its characteristics. A mutation in the germ cells of sexually reproducing organisms may be transmitted to the offspring, whereas a mutation that occurs in somatic cells may be only transferred to descendent daughter cells.

Nanogram (ng)
One nanogram is 1 x 10^{-9} gram. There are 1,000,000,000 ng in one gram.

Nanoparticles
These are usually considered to be particles of less than 100 nanometres diameter. One nanometre is a millionth of a mm. To put into some context this is about a ten thousandth of the width of a human hair.

2-naphthylamine
A chemical used in the past in the manufacture of dyes. It is made up from 2 benzene rings with a nitrogen and hydrogen side chain.

Non-Hodgkin lymphoma
A type of malignant cancer of the lymphatic system or lymphoid tissue. Most lymphoma are of this type (as opposed to being Hodgkin lymphoma).

Number concentration of particles
The number of particles found in a specified volume of air, usually 1 cubic metre.

Pharynx
The throat and back of the nose.

Point sources
Sources of pollution from a fixed point in space e.g. an industrial site. The term is used in contrast to mobile sources of pollution e.g. cars.

Polycyclic aromatic hydrocarbons (PAHs)
These are a group of structurally related organic compounds that contain 2 or more fused rings. They are formed as a result of combustion/pyrolysis.

PM_{10}, $PM_{2.5}$
The concentration (expressed in µg/m^3) of particles generally less than 10µm and 2.5µm respectively[6]. The terms PM_{10} and $PM_{2.5}$ are sometimes used to describe particles of diameter of less than 10 and 2.5 µm respectively but this is not strictly correct: the terms refer to the concentrations of particles and not to the particles themselves.

Picogram (pg)
A picogram is 1x 10^{-12} gram. There are 1,000,000,000,000 pg in one gram.

[6] Strictly, particles that pass a sampler entry with 50% efficiency at 10 micrometres or 2.5 micrometres respectively.

Spontaneous mutation
A mutation that occurs as a result of natural processes in cells, as opposed to those that arise because of interaction with an outside agent or mutagen.

Soft tissue sarcomas
These are a rare type of cancer that develop from cells in the soft, supporting tissues of the body such as muscle, fat and blood vessels. They may occur in limbs, chest, abdomen or pelvis and less commonly in head and neck.

TCDD
The most studied dioxin, and the one that is used as a reference compound when considering the toxicity of mixtures of dioxins, is often referred to simply as TCDD. This is an abbreviation of its full chemical name, 2,3,7,8-tetrachlorodibenzo-p-dioxin. It is considered the most toxic dioxin.

TEOM
Tapered Element Oscillating Micro-balance. An instrument used to measure the mass concentration of particles in the air. Particles are collected on a vibrating rod: the mass deposited affects the frequency of vibration of the rod and this, being recorded, allows the mass of particles in the air to be calculated.

Tolerable Daily Intake (TDI)
An estimate of the amount of contaminant, expressed on a body weight basis (e.g., mg/kg body weight) that can be ingested daily over a lifetime without appreciable health risk.

Total suspended particulates
A measure of particles derived by collecting particles of approximately 100 µm or less in a sampler. This includes particles that are too large to enter the lung. The measurement method has generally been superseded by measurement of PM_{10}.

Toxic Equivalency Factor (TEF)
A measure of the relative toxicological potency of a chemical compared to a well characterised reference compound. TEFs can be used to sum the toxicological potency of a mixture of chemicals which are all members of the same chemical class, having common structural, toxicological and biochemical properties e.g. dioxins. In the case of dioxins the reference compound is TCDD.

Toxic Equivalent (TEQ)
This is a method of comparing the total relative toxicological potency within a mixture using TEFs (see above). It is calculated as the sum of the products of the concentration of each chemical multiplied by the TEF.

Ultrafine component
The component of particles less than about 100 nm in diameter.

Uncertainty factors
Value used in extrapolation from experimental animals to man (assuming that man may be more sensitive) or from selected individuals to the general population; for example, a value applied to the No Observed Adverse Effect Level (NOAEL) to derive a TDI. The value depends on the size and type of population to be protected and the quality of the toxicological information available.

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Using metal ratios to detect emissions from municipal waste incinerators in ambient air pollution data

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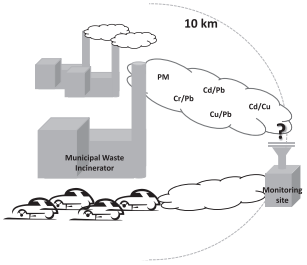
HIGHLIGHTS

- Metal ratios used to fingerprint emissions from UK municipal waste incinerators.
- Weekly ambient metals data and high-resolved met data were used.
- No evidence of incinerator emissions within 10 km around four installations.
- Ambient metal ratios agreeing with emissions in sites within 10 km of two plants.
- Plume grounding detected for less than 0.2% of the time, contributing little to PM.

GRAPHICAL ABSTRACT



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ABSTRACT

This study aimed to fingerprint emissions from six municipal waste incinerators (MWIs) and then test if these fingerprint ratios could be found in ambient air samples. Stack emissions tests from MWIs comprised As, Cd, Cr, Cu, Pb, Mn, Ni, V and Hg. Those pairs of metals showing good correlation ($R > 0.75$) were taken as tracers of MWI emissions and ratios calculated: Cu/Pb; Cd/Pb; Cd/Cu and Cr/Pb. Emissions ratios from MWIs differed significantly from those in ambient rural locations and those close to traffic. In order to identify MWI emissions in ambient air two analysis tests were carried out. The first, aimed to explore if MWI emissions dominate the ambient concentrations. The mean ambient ratio of each of the four metal ratios were calculated for six ambient sampling sites within 10 km from a MWI under stable meteorological conditions when the wind blew from the direction of the incinerator. Under these meteorological conditions ambient Cd/Pb was within the range of MWI emissions at one location, two monitoring sites measured mean Cr/Pb ratios representative of the MWI emissions and the four sites measured values of Cu/Pb within the range of MWI emissions. No ambient measurements had mean Cd/Cu ratios within the MWI values. Even though MWI was not the main source determining the ambient metal ratios, possible occasional plume grounding might have occurred. The second test then examined possible plume grounding by identifying the periods when all metal ratios differed from rural and traffic

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values at the same time and were consistent with MWI emissions. Metal ratios consistent with MWI emissions were found in ambient air within 10 km of one MWI for about 0.2% of study period. Emissions consistent with a second MWI were similarly detected at two ambient measurement sites about 0.1% and 0.02% of the time. Where plume grounding was detected, the maximum annual mean particulate matter (PM) from the MWI was estimated to be 0.03 μg m^{-3} to 0.12 μg m^{-3}; 2–3 orders of magnitude smaller than background ambient PM_{10} concentrations. Ambient concentrations of Cr increased by 1.6–3.0 times when MWI emissions were detected. From our analysis we found no evidence of incinerator emissions in ambient metal concentrations around four UK MWIs. The six UK MWIs studied contributed little to ambient PM_{10} concentrations.

1. Introduction

Municipal waste consists of a mix of combustible and non-combustible materials such as paper, plastic, food waste, glass, defunct household appliances and other non-hazardous materials (EMEP- EEA, 2013) that might vary by time and by city, town or village. The use of Municipal Solid Waste Incinerators (MWIs) has been increasing in the United Kingdom (UK) as a means to treat municipal waste due to European Union (EU) restrictions on the use of landfills. Modern European MWIs have operated under the EU Waste Incineration Directive (EU-WID) 2000/76/EC which set limits on emissions for heavy metals, dioxins and furans, carbon monoxide, dust, total organic carbon, hydrogen chloride, hydrogen fluoride, sulphur dioxide and nitrogen oxides. The EU-WID came into operation in 2002 for new MWIs and applied to all existent ones from 2005. The later Directive on Industrial Emissions (IED) (2010/75/EU) merged seven directives, including the EU-WID, into one piece of legislation, in order to harmonise the various strands of industrial regulation. The implementation of the IED in the UK was set to 2013 for new installations and 2014 for the existing ones.

Despite the strict limits on emissions, there is still considerable public concern about possible health effects associated with incineration. Some epidemiological studies have reported significant positive relationships with broad groups of congenital anomalies in populations living near MWIs. However, the results from these studies remain inconclusive due to limitations on exposure assessment, possible confounding risk factors and lack of statistical power (Ashworth et al., 2014).

Previous studies found no evidence that incinerators had a major or modest impact on particulate concentrations either in the United States (Shy et al., 1995) or in the UK (Ashworth et al., 2013). Despite this, older MWI have been found to be a source of heavy metals to the atmosphere (Sakata et al., 2000; Hu et al., 2003; Moffet et al., 2008) and high concentrations could be found in soil and vegetation samples in the vicinities of MWIs (Morselli et al., 2002).

In this study we aimed to fingerprint emissions from UK MWIs by identifying characteristic metal emission ratios and then test if these fingerprint ratios can be found in ambient air samples around MWI. Our analysis was part of a UK Public Health England (PHE) project investigating birth outcomes in the population living around (10 km distance) MWIs in England, Wales and Scotland.

2. Methods

2.1. Metals emissions from MWI and ambient concentrations

Quarterly stack emissions tests from MWIs were made available by the UK Environment Agency (EA). Particulate matter was sampled isokinetically from each MWI stack onto quartz filters. Following acid digestion with a mixture of nitric and hydrofluoric acid, stack samples were analysed by Inductively Coupled Plasma - Mass Spectrometry (ICP-MS) according to EN 14385:2004. This method is validated against matrix reference material BCR-037. Samples were analysed for Arsenic (As), Cadmium (Cd), Chromium (Cr), Copper (Cu), Lead (Pb), Manganese (Mn), Nickel (Ni), Vanadium (V) and Mercury (Hg). Emissions data were available from 2003 until 2010. For most of the samples the metals concentrations were aggregated for reporting purposes (EU-WID compliance) and the concentration for each metal was not available. Only 52 tests among all the UK MWI had detailed concentration values for each metal and these were used for fingerprinting. This detailed metals emissions data came from 10 (of a total of 22) UK MWIs: Crymlyn Burrows, Chineham, Coventry, SELCHP, Dudley, Bolton, Stockton-on-Tees, Stroke-on-Trent, Tyseley and Wolverhampton.

Ambient concentrations of As, Cd, Cr, Cu, Iron (Fe), Hg, Mn, Ni, Pb, Platinum (Pt), V and Zinc (Zn) in PM_{10} (particulate matter with aerodynamic diameter <10 μm) were measured by sampling onto a filter (cellulose ester) for weekly periods using a Partisol 2000 sampler according to EN12341:2014. The ambient air filters were digested in a mixture of nitric acid and hydrogen peroxide in a microwave oven, according to EN 14902:2005, and followed by analysis by ICP-MS. This procedure was validated by the digestion and measurement of suitable matrix reference materials, such as NIST SRM 1648a – urban particulate matter. The recoveries of all relevant metals were consistent with the certified values within the uncertainty of the measurements. The analysis was undertaken by the National Physical Laboratory (NPL) for sites belonging to the urban and industrial metals network; and by the Centre for Ecology and Hydrology (CEH) for the rural metals network. These data are available as monthly means at http://uk-air.defra.gov.uk/.

Due to high sampling temperatures the stack filters are quartz and a hydrofluoric (HF) acid matrix is required to digest them to ensure that any deeply trapped PM is recovered, and to perform an appropriate blank correction. By contrast only nitric acid digestion is needed to fully digest cellulose ester filters used for ambient measurements and HF digestion is not required. Kulkarni et al. (2007) underlined the importance of HF digest for ambient PM samples with high silica mineral content. This was unlikely to be an issue in our study since large mineral particle emissions from the MWI would have been preferentially trapped in the bag filtration system that have higher efficiency for larger particles (Buonanno et al., 2009; Ashworth et al., 2013) and mineral dust episodes such as those from the Sahara are rare in the UK (Ryall et al., 2002).

In order to fingerprint emissions from MWIs, the correlation coefficient between metals was calculated from the stack measurements. Those pairs of metals showing good correlation ($R > 0.75$) across all MWI sites were taken as potential tracers for MWI emissions. Ratios were then calculated by means of Reduced Major Axis (RMA) regression (Ayres, 2001; Warton et al., 2006). Due to insufficient samples it was not possible to create fingerprint

profiles for individual MWI.

Ratios for the same metals were calculated from ambient samples from the rural network (n = 579 samples from 11 sites in 2010) and from Cromwell Road site in London as representative of metal ratios from traffic sources (data from 2004 to 2011, n = 311).

2.2. Detecting MWI emissions in ambient air

Six metals sampling sites were located within 10 km of a MWI in the UK (Fig. 1; Table 1) with weekly samples of ambient metals concentrations. Most of the ambient metals sampling sites were located close to heavily industrialized areas. The sampling sites Walsall Bilston Lane (Background metals site) and Walsall Centre (Industrial metals site), near the Wolverhampton and Dudley MWIs respectively, had multiple industries related to metals refining and finishing located nearby. Although the Redcar Normanby site was an urban background site, the same wind direction towards the Stockton-on-Tees MWI included industrial premises such as chemical, plastics and acrylics manufacturers and an oil refinery. London Westminster and Sheffield Centre were urban background sites located near traffic. London Westminster site had no industrial sources nearby. NE of the Sheffield Centre metals site (in the same direction as the MWI) there were several industries producing industrial alloys, cast products and steel. The Swansea Morriston sampling site was located just off a main road running SW – NE. The Crymlyn Burrows MWI was located SE of the metals site with the UK's largest steel production plant (Port Talbot) located ~3 km to the east of the MWI.

The analysis period for each MWI was determined by the operational times for each MWI and the coincidental availability of ambient metal measurements (Table 1).

To assess if emissions from MWIs were detected at the metals sites, two sets of analysis were undertaken: the first aimed to explore if MWI emissions dominate the ambient concentrations; the second tested if all four ratios differed from rural or traffic values at the same time and were consistent with MWI emissions. For the first analysis, bivariate polar plots (BPP) of those metal ratios that were identified as good tracers of MWI emissions were calculated using the Openair R-package (Carslaw and Ropkins, 2012). BPP determine the mean value of an ambient metals ratio against wind direction and wind speed. BPP have been used previously in receptor analysis to identify the location of potential sources of air pollution (Carslaw, 2005). For the second analysis, the Polar Annulus (PA) function of the Openair R-package was calculated. PA plots show the time series of the measured ambient ratio by wind direction.

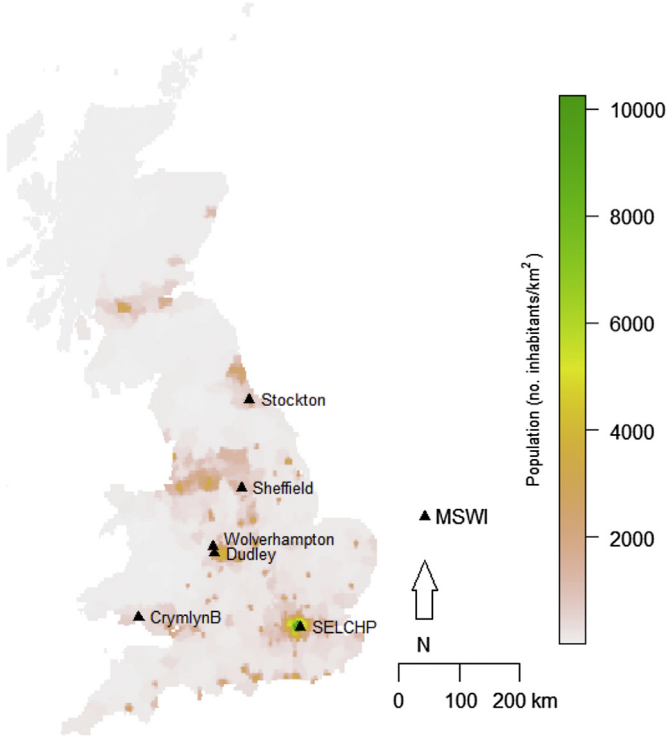

Fig. 1. Map of the UK MWI included in this study. Base map: population density in 2000 (CIESIN, 2011).

Previous studies have successfully determined the sources and their contribution to pollutant concentrations measured at low-frequency (e.g. daily, weekly). Different techniques have been proposed in the literature based on the frequency of the wind for a given wind sector (e.g. Cosemans et al., 2008; Godri et al., 2010). Here, high resolution (hourly) meteorological measurements were used to compute BPP and PA. The same weekly metals concentration was assigned to each hourly measurement of wind speed and direction.

Meteorological data were obtained from weather stations within 30 km of a MWI and processed using the Atmospheric Dispersion Modelling System Urban (ADMS-Urban). ADMS-Urban uses meteorological variables including wind speed, wind direction, temperature and cloud cover to calculate parameters that are used in the dispersion algorithms such as boundary layer height, Monin-Obukhov length, etc. The meteorological input data was extracted from the closest weather station following the MetOffice quality standards. Missing cloud cover records were completed using data from the nearest met station with 90% completeness where necessary.

During unstable meteorological conditions buoyant motions are enhanced causing rapid dispersion of emissions. Under poor mixing (stable) conditions MWI plume will mix less with the surrounding air keeping its chemical composition. Highest concentrations attributable to the MWI at the ambient metal sites are therefore expected under stable atmospheric conditions. For this reason, analysis of ambient data was focused on the times when stable atmospheric conditions were met.

Atmospheric stability conditions were defined as:

$$z^{*}\mathrm{L} = z^{*}1/\mathrm{L_{MO}}$$

where z is the boundary layer height and $\mathrm{L_{MO}}$ is the Monin-Obukhov length. Stability conditions were classified as unstable for $-1000 < z^{*}\mathrm{L_{MO}} \leq -0.2$; neutral for $-0.2 < z^{*}\mathrm{L_{MO}} \leq -0.2$; and stable for $0.2 < z^{*}\mathrm{L_{MO}} \leq 50$.

During stable conditions an elevated point source (such as the MWI chimney) may be above the boundary layer height (z). At these times emissions can be released above the temperature inversion and hence not influence ground-level concentrations. With the exception of SELCHP MWI, z was lower than the MWI chimneys at all times. At SELCHP, the stack was above the boundary layer for 22% of the hours when stable conditions were met.

$\mathrm{NO_X}$, $\mathrm{SO_2}$ and PM emissions reported by the MWI operators were used to assess the days when the plants were operating for inclusion in the BPP and PA calculations. The total analysis period covered 11,437 days.

In order to test our assumptions about higher attributable MWI emissions under stable conditions and whether emissions from stacks could be identified using weekly samples instead of highly-resolved data (i.e. hourly concentrations), two sensitivity tests were undertaken. These used air quality data from the Harwell monitoring site (1.3265°W, 51.5711°N). Harwell is a rural monitoring site

Table 1
List of MWI and ambient metals sites (AMS) within 10 km.

MWI	Stack height (m)	Start year	Median PM emissions (kg day^{-1})	Ambient metals site (AMS)	Type of AMS	Distance MWI to AMS (km)	Analysis period (no. days)
Crymlyn Burrows	40	2003	0.7	Swansea Morriston	Traffic	5.2	847
Dudley	47	1998	1.8	Walsall Bilston Lane	Background	9.7	1578
Dudley	47	1998	1.8	Walsall Centre	Industrial	10.3	1515
Stockton-on-Tees	70	1998	4.5	Redcar Normanby	Background	9.1	745
Sheffield	76	1990	0.7	Sheffield Centre	Background	1.9	910
SELCHP	100	1994	14.8	London Westminster	Background	6.0	1889
Wolverhampton	76	1998	3.0	Walsall Bilston Lane	Background	5.8	1996
Wolverhampton	76	1998	3.0	Walsall Centre	Industrial	8.1	1957

belonging to the UK Automatic Urban and Rural Network (AURN) located 7.3 km from the coal-fired Didcot Power Station, a well known source of atmospheric SO_2 (McGonigle et al., 2004; Charron et al., 2005). The first test was based on BPP for SO_2 concentrations under unstable, neutral and stable conditions. BPP for SO_2 concentrations under stable conditions showed higher concentrations in the direction of the Power Station compared to those measured under unstable conditions (Supplementary Figure A1). Moreover, under unstable conditions, the source of SO_2 in the direction of the Power Station was spread over a wider range of wind sectors and wind speeds due to enhanced atmospheric mixing conditions. In the second test, BPP and PA for weekly SO_2 concentrations were compared to those using hourly data. Under stable conditions, using weekly mean SO_2 with hourly resolved meteorological data, BBP analysis located the same source of SO_2 as using hourly data (Supplementary Figure A2). The PA time series of the trends of the SO_2 source computed from the hourly and weekly datasets were also similar (Supplementary Figure A3). The use of weekly means combined with high-resolved meteorological data can therefore be confidently used to detect point sources of atmospheric pollution and to assess the temporal changes in their intensity.

2.3. Quantification of ambient MWI PM using a single metals tracer

Analysis of ambient metal ratios can be used to detect MWI emissions but not quantify their impacts on ambient PM concentrations. To quantify Particulate Matter (PM) at receptor locations the ratio of PM/metal emitted by the MWIs was calculated from stack emissions tests by RMA regression.

ADMS-Urban was also used to model daily mean PM concentrations at post-code resolution for each MWI following the methods detailed in Ashworth et al. (2013). Metals concentrations were then estimated at the receptor (ambient metal site) based on modelled PM and calculated stack emission ratios.

3. Results

3.1. Metals emissions from MWI

The MWI listed in Table 1 were installations that were adapted to the EU-WID except Crymlyn Burrows which was commissioned following EU-WID. Results from MWI stack tests are summarized in Table 2. The ambient concentrations measured at rural metals sites are also given for comparison. Sorted from largest to smallest emissions concentrations (median values), MWI were emitters of Pb > Cr > Ni > Mn > Cu > Cd > As > V. When compared to the median rural background concentrations, MWI emissions contained greater quantities of Cr (41×10^3 times larger than rural concentrations), Cd (22×10^3 times larger), Ni (13×10^3), Pb (5×10^3) and Cu (3×10^3).

Emissions of Cu–Pb showed a good correlation ($R = 0.91$, $N = 18$, $p < 0.001$) followed by Cd/Pb ($R = 0.86$, $N = 19$, $p < 0.001$), Cd/Cu ($R = 0.77$, $N = 50$, $p < 0.001$) and Cr/Pb ($R = 0.68$, $N = 19$, $p < 0.05$) (Table 3). We therefore selected the emissions ratios of these four pairs of metals as potential tracers for MWI emissions. Additionally their values also differed from those found in rural and traffic locations (Table 4). However, these four ratios represented only three pieces of independent information since (Cd/Pb)/(Cd/Cu) = (Cu/Pb).

Since Cr and Cd were the two metals with the greatest enrichment factors (Table 2), ratios of PM/Cd and PM/Cr were calculated from emissions samples. Emissions rates of PM/Cd showed a good correlation coefficient ($R = 0.94$, $p < 0.01$, $n = 34$) while PM/Cr was weaker ($R = 0.41$, $p < 0.05$, $n = 33$). Cd therefore was therefore additionally selected as tracer for PM emitted by MWI using a ratio of 6724 [5999–7647, 2σ] mgPM (mgCd)$^{-1}$ (Table 5).

3.2. Detecting MWI emissions in ambient air

Fig. 2a shows an example of BPP for the Cr/Pb ratio calculated using weekly samples from the Redcar Normanby site, 9.1 km from the Stockton-on-Tees incinerator. Values of Cr/Pb fell within the range of MWI emissions for wind speeds higher than 10 m s^{-1} when the wind blew from the direction of the incinerator (Fig. 2b). For wind speeds lower than 10 m s^{-1}, values of Cr/Pb ranged between that found in rural areas and the ratio expected from traffic sources.

Ambient ratios measured at the metal sites from the wind direction of each MWI were compared with the values from MWI stacks (Table 6). With the sole exception of the Walsall Centre sampling site in the direction of Wolverhampton incinerator, mean Cd/Pb, mean Cd/Cu and mean Cd/Pb were not within the range of MWI emissions for any of the ambient measurements near a MWI. However, near the Dudley, Stockton-on-Tees and Wolverhampton MWI ambient ratios were above the rural background values for Cd/Cu; and above the traffic values for Cd/Pb near Stockton-on-Tees and Wolverhampton. The metals sites near Stockton-on-Tees and Sheffield measured Cr/Pb ratios representative of the MWI emissions. The values of the Cu/Pb ratio near Dudley, Stockton-on-Tees, Sheffield and Wolverhampton fell within the range of values representative of the MWI emissions.

From analysis of the mean ambient metal ratios when the wind blew from the direction of an incinerator it was clear that the MWIs were not the main source of the tracer metals. Only Redcar Normanby measured mean ratios within the MWI emissions range (Cr/Pb and Cu/Pb) and different from the rural (Cd/Cu) and traffic values (Cd/Pb). However, emissions from MWI might influence the burden of metals in the area and grounding of MWI plume might have occasionally occurred. This would lead to a mixture of sources on a weekly filter producing a ratio that was between rural conditions and the incinerator emissions. Fig. 3 shows the time series of the Cr/

Table 2
Minimum, mean and maximum metal concentrations from MWI stack tests from 2003 to 2010. Ambient concentrations measured at rural metals sites in 2010 are also shown.

	MWI stack emissions (2003–2010)				Ambient rural concentration (2010)			
Metal	Min (μg m^{-3})	Median (μg m^{-3})	Max (μg m^{-3})	*N*	Min (ng m^{-3})	Median (ng m^{-3})	Max (ng m^{-3})	*N*
As	0.00	0.85	97.00	50	0.05	0.40	13.6	579
Cd	0.00	1.30	26.50	52	0.01	0.06	2.05	579
Cr	0.00	10.60	94.00	51	0.24	0.26	7.06	579
Cu	1.00	6.10	160.00	50	0.12	1.98	60.3	579
Pb	0.00	16.00	200.00	19	0.36	3.38	184	579
Mn	0.40	6.30	92.30	52	0.04	1.55	52.1	579
Ni	0.00	6.80	177.50	49	0.06	0.52	9.74	579
V	0.00	0.75	12.20	49	0.12	0.74	11.2	579

Table 3
Correlation coefficients between the metals emitted by the MWI in the UK from 2003 to 2010.

	As	Cd	Cr	Cu	Pb	Mn	Ni	V
As	1							
Cd	−0.05	1						
Cr	0.08	0.36	1					
Cu	0.00	**0.77**	0.57	1				
Pb	0.03	**0.86**	**0.68**	**0.91**	1			
Mn	0.02	0.42	0.13	0.39	0.42	1		
Ni	−0.06	0.06	0.03	0.06	−0.04	0.24	1	
V	0.11	0.40	0.02	0.20	−0.05	0.42	0.05	1

Pb ratios measured at the Swansea Morriston metals site when the wind blew from Crymlyn Burrows MWI. Despite the Cr/Pb mean ratio being between the rural and traffic values (Table 6), some of the measured ratios measured fell within the MWI emissions range (Fig. 3).

Most of the MWI are located in areas of diverse industrial sources which could confound the analysis. Even though an ambient metal ratio was within the range of MWI emissions (Fig. 3), we could not with certainty attribute ambient metals concentrations to direct emissions from MWIs. It was clearly not possible to fingerprint all of the potential emissions sources. We therefore

Table 4
Mean and 95% confidence interval of the metals ratio values representative of MWI emissions, ambient rural and ambient traffic locations.

Metals ratio	MWI (mean ± 2σ)	Rural (mean ± 2σ)	Traffic (mean ± 2σ)
Cd/Cu	0.14 [0.12–0.17]	(0.26 [0.24–0.28])·10^{-1}	(0.07 [0.06–0.08])·10^{-1}
Cd/Pb	0.08 [0.06–0.10]	(1.31 [1.27–1.35])·10^{-2}	(0.17 [0.16–0.18])·10^{-1}
Cr/Pb	0.56 [0.38–0.75]	0.13 [0.12–0.14]	0.28 [0.25–0.31]
Cu/Pb	0.83 [0.67–0.99]	0.51 [0.47–0.54]	2.38 [2.14–2.63]

Table 5
Mean and 95% confidence interval of the PM_{10}/Cd and PM_{10}/Cr ratios representative of MWI emissions.

	Cd	Cr
PM_{10}/metal (mean ± 2σ)	6724 [5999–7647]	1708 [1166–2249]
R	0.94	0.41
N	34	33

considered the time series of the four ratios measured at each site to determine whether all the metal ratios were consistent with MWI emissions or were different from rural or traffic values at the same time. Fig. 4 shows the time series of the four metal ratios measured at Redcar Normanby from the direction of the Stockton-on-Tees incinerator. Measurements of Cr/Pb were within the range of MWI emissions on three occasions and these were coincidental

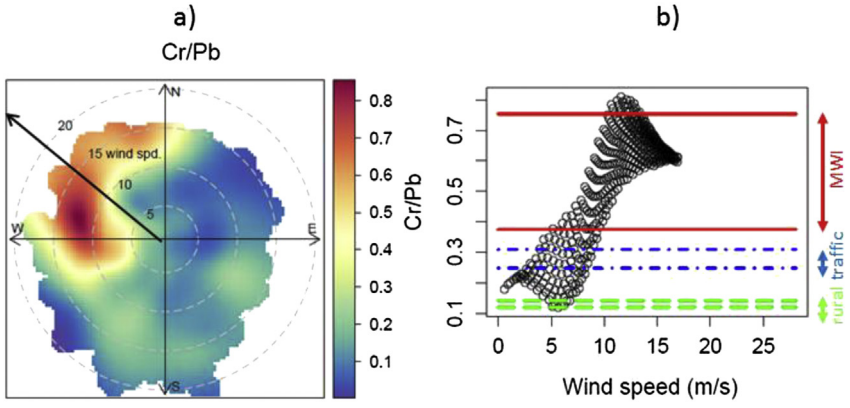

Fig. 2. (a) BPP for the Cr/Pb ratio measured at Redcar Normanby metals site under stable atmospheric conditions. Radially wind direction is plotted from north (N). The concentric lines indicate increasing intensity of the wind speed and the shading shows the mean ratio value. The arrow indicates the direction where the Stockton-on-Tees MWI is located. (b) Distribution of the ratio values against wind speed for the direction where the Stockton-on-Tees MWI is located. Solid red horizontal lines indicate the range of MWI emissions for the Cr/Pb ratio; green and blue dashed lines indicate the range for the rural and traffic, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 6
Minimum and maximum ambient values for the four metal ratios measured in the ambient metals sites near a MWI when the wind blew from the incinerator (30° sectors). Numbers in italics indicate ratios that are different from rural or traffic ratios. * indicates ratios were within the MWI emissions representative values.

Metals site	MWI	Cd/Cu	Cd/Pb	Cr/Pb	Cu/Pb
Swansea Morriston	Crymlyn Burrows	0.01–0.02	0.01–0.02	0.03–0.27	1.21–1.79
Walsall Bilston Lane	Dudley	*0.04–0.06*	*0.03–0.06*	0.03–0.10	0.73–1.47*
Walsall Centre	Dudley	*0.02–0.05*	*0.02–0.04*	0.08–0.17	0.70–0.93*
Redcar Normanby	Stockton-on-Tees	*0.03–0.05*	*0.02–0.02*	*0.12–0.81**	0.49–0.82*
Sheffield Centre	Sheffield	0.01–0.02	0.01–0.02	*0.02–0.51**	0.89–1.93*
London Westminster	SELCHP	0.01–0.02	0.01–0.03	0.14–0.34	1.32–1.80
Walsall Bilston Lane	Wolverhampton	*0.03–0.06*	*0.04–0.06*	0.06–0.19	0.78–1.91*
Walsall Centre	Wolverhampton	*0.02–0.10*	*0.02–0.08**	0.08–0.20	0.65–0.99*

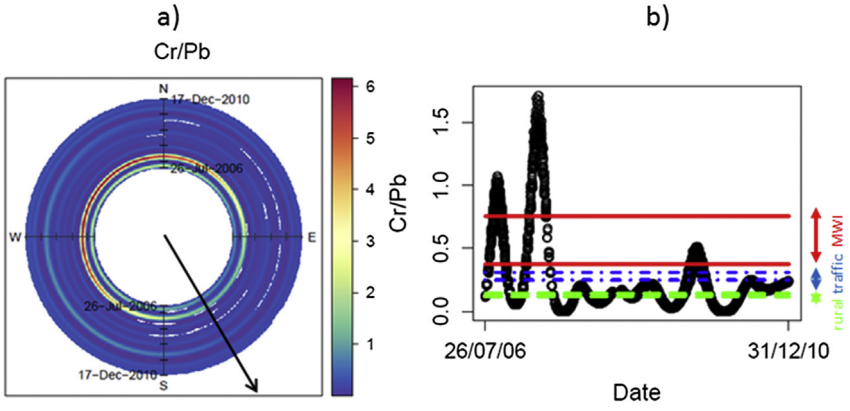

Fig. 3. (a) PA for the Cr/Pb ratio measured at Swansea Morriston metals site under stable atmospheric conditions. The arrow indicates the direction where the Crymlyn Burrows MWI is located. (b) Time series of the Cr/Pb ratio at the direction where the Crymlyn Burrows MWI is located. Solid horizontal red lines indicate the range of MWI emissions for the Cr/Pb ratio; green and blue dashed lines indicate the range for the rural and traffic representative values, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

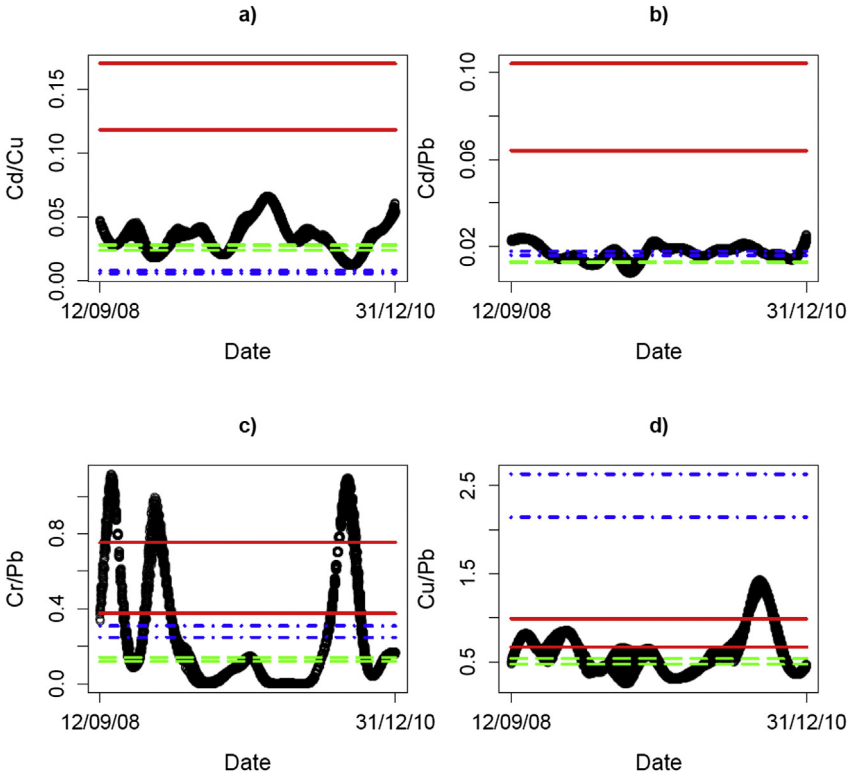

Fig. 4. Time series of Cd/Cu (a), Cd/Pb (b), Cr/Pb (c) and Cu/Pb (d) measured at Redcar Normanby when the wind blew from the direction where the Stockton-on-Tees MWI is located. Solid horizontal red lines indicate the range of MWI emissions; green and blue dashed lines indicate the range for the rural and traffic representative values, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

with Cu/Pb values expected from MWI emissions. Measured Cd/Cu ratios were similar to the rural concentrations but some peaks were different from rural values indicating the presence of a source changing the metals ratio that might have been the incinerator. Similarly, Cd/Pb ratios ranged between the rural and traffic values with some peaks moving towards the MWI values.

A further example of this approach is shown in Fig. 5 which depicts the time series of the metal ratios measured at London Westminster for the wind sector where SELCHP MWI is located. Some peaks of the Cr/Pb ratio fell within the MWI ratio (Fig. 5c) but those peaks were identified as traffic sources by the other tracers, which was consistent with this being a highly trafficked location in central London, leading to the conclusion that these peaks were not due to the MWI plume grounding.

For the majority of the metals sampling sites located near an incinerator it was rare to measure simultaneously the four ratios within MWI values or different from the rural and traffic values (Table 7). Only the Redcar Normanby, Walsall Centre and Walsall Bilston Lane metals sites measured all four ratios coincidently; for 5.4%, 2.6% and 0.5% of the time, respectively, that the wind blew from the incinerator Stockton-on-Tees (Redcar Normanby) and Wolverhampton (Walsall Centre and Walsall Bilston Lane) in stable conditions.

For the metals sampling sites near Stockton and Wolverhampton MWIs, PM emitted by the MWI was estimated using the Cd concentration measured when grounding of the MWI plume was detected by all four tracer ratios; and assuming zero PM concentration from MWI when plume grounding was not detected. It is therefore an estimated maximum PM from MWI at the ambient metals site. The maximum PM concentration in ambient air from MWI emissions ranged from 0.029 μg m^{-3} (Stockton-on-Tees MWI) to 0.123 μg m^{-3} (Wolverhampton MWI). This differed from the mean PM concentrations predicted from the ADMS-Urban model that were 1–2 orders of magnitude smaller (Table 8).

The ambient concentration of heavy metals during plume grounding was compared with the mean concentration measured from other wind sectors (Table 9). The concentrations of Cr was higher at all measurement sites when MWI emissions were detected: 3.0 (at Redcar Normanby), 1.6 (Walsall Bilston Lane) and 2.4 (Walsall Centre) times larger compared with the mean concentration measured from the other wind sectors. Ni concentrations were higher at Walsall Bilston Lane when MWI emissions from Wolverhampton were detected (14.3 ± 2.3 ng m^{-3}) compared to the mean concentration measured from the other wind sectors (4.6 ± 5.8 ng m^{-3}). The other metals sites (Redcar Normanby and Walsall Centre) also measured higher mean concentrations but differences were not statistically significant.

The emission ratio in Table 5 can be combined with the ADMS-Urban PM estimates to calculate a metal concentration at each ambient metals site, as shown in Supplementary Material (Part B). This shows that the contribution of MWI emissions to the ambient levels of Cd and Cr were very small (ranging from 0.001% to 0.08%).

4. Discussion and conclusions

In our study we aimed to pin-point emissions from MWI using measurements of ambient heavy metal particle concentrations. Several studies have used receptor models to apportion particulate matter sources or to apportion bulk deposition near a MWI (Venturini et al., 2013). Receptor models are useful when the aim of the study is to identify the sources of pollution affecting an ambient measurement site. However, in our study we aimed to pin-point only one source of pollution (incinerator) instead of explaining all sources influencing the measured metals concentrations.

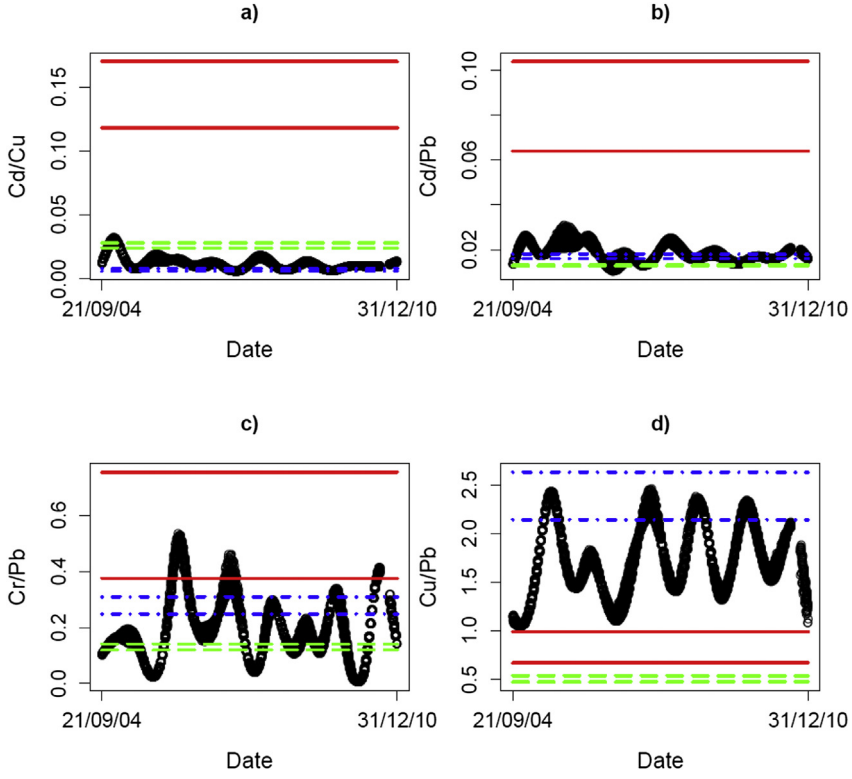

Fig. 5. Time series Cd/Cu (a), Cd/Pb (b), Cr/Pb (c) and Cu/Pb (d) ratios measured at London Westminster at the direction where the SELCHP MWI is located. Solid horizontal red lines indicate the range of MWI emissions; green and blue dashed lines indicate the range for the rural and traffic representative values, respectively. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 7
Percentage of time that the four tracers were simultaneously within the range of MWI emissions or different from rural and traffic ambient sources.

MWI	Ambient metals site	% time with wind from MWI under stable conditions	% of the study period
Crymlyn Burrows	Swansea Morriston	0.0	0.0
Dudley	Walsall Bilston Lane	0.0	0.0
Dudley	Walsall Centre	0.0	0.0
Stockton-on-Tees	Redcar Normanby	5.4	0.2
Sheffield	Sheffield Centre	0.0	0.0
SELCHP	London Westminster	0.0	0.0
Wolverhampton	Walsall Bilston Lane	0.5	0.0
Wolverhampton	Walsall Centre	2.6	0.1

Table 8
Estimated PM from MWIs at ambient metals sites.

MWI	Ambient metals site	Maximum PM from MWI ambient data (mean ± σ) (μg m^{-3})	PM from MWI ADMS-Urban (mean ± σ) (μg m^{-3})
Stockton-on-Tees	Redcar Normanby	0.029 ± 0.002	$0.008 \cdot 10^{-1}$ ± 0.002
Wolverhampton	Walsall Bilston Lane	0.038 ± 0.002	0.002 ± 0.003
Wolverhampton	Walsall Centre	0.123 ± 0.007	0.001 ± 0.002

First, we successfully fingerprinted emissions from modern MWI in the UK using stack emissions samples of heavy metals. The ratios used to fingerprint MWI emissions in UK were consistent with emissions from burning electronic waste (that emits Cu and Pb) (Gullett et al., 2007), mixed paper and plastics (which emit Pb and Cd), and batteries (which emit Cd; Hasselriis and Licata, 1996; WHO, 2010), materials all expected to be found in municipal waste. Cr is emitted when burning coloured newsprint and mixed paper, plastic film, lawn waste, wood, textiles, footware and fines (Hasselriis and Licata, 1996). It has not been previously used as a tracer of MWI emissions despite being emitted in high abundance relative to rural concentrations (Table 2).

Unfortunately the number of stack samples available to calculate emission ratios was not enough to calculate individual source profiles for each MWI or to assess their changes over time. However, emission ratios of heavy metals were expected to be consistent between MWI for two reasons. First, the metals with best correlation coefficients share common origins within waste material. In a previous study from a MWI in British Columbia found that many waste types contributing to Pb emissions also exhibited high levels of Cr. Garden waste and certain type of paper fractions (commonly found in municipal waste) contain the highest concentrations of Pb, Cr and Cd. Emissions of Cr and Cd versus Pb also showed a linear relationship (Hasselriis and Licata, 1996). This is in agreement with our results. Second, all MWI in England and Wales used the same abatement techniques for heavy metals. These include injection of activated carbon (to capture mercury) and bag filters (to remove particulates). Furthermore the Cd/Pb ratio in our study was almost identical to that reported for modern European MWI in Nielsen et al. (2010) (Table 10). Other ratios were more similar to Nielsen et al. (2010) values than the older studies of Morselli et al. (2002) and Hu et al. (2003). Although it did not affect our study, the revised Restriction of the use of certain Hazardous Substances (RoHS) directive (2011/65/EU), that became effective on January 2013, limits the use of hazardous substances (such as Pb, Hg, Cd, and Cr (VI), among other substances) in electrical and electronic equipment. Emissions of heavy metals from incinerators are therefore expected to decrease and this will impact on future emission ratios.

In order to properly detect sources of atmospheric pollution in ambient data three requirements are needed: i) the dataset collected must include daily, seasonal and yearly variations of the source (Cohen et al., 2014); ii) the emissions tracers are not transformed in the atmosphere between emission and detection; and iii) the ambient measurements should include the chemical species emitted by the source (Cohen et al., 2014) at a measurable concentration.

The stack emissions used to fingerprint MWI emissions

Table 9
Mean ± standard deviation of heavy metals concentration (ng m^{-3}) measured at the ambient metals site when MWI emissions were detected and for the other wind directions. Bold numbers indicate those heavy metals that concentrations (within 95% confidence) were higher when MWI plume was detected.

	Redcar Normanby (Stockton-on-Tees)	Walsall Bilston Lane (Wolverhampton)	Walsall Centre (Wolverhampton)
As MWI plume	0.20 ± 0.03	0.88 ± 0.06	1.17 ± 0.08
As other sectors	0.40 ± 0.21	1.17 ± 0.38	1.05 ± 0.34
Cd MWI plume	0.08 ± 0.01	1.17 ± 0.06	0.71 ± 0.03
Cd other sectors	0.09 ± 0.04	2.58 ± 1.30	0.61 ± 0.40
Cr MWI plume	**2.32 ± 0.80**	**5.84 ± 0.21**	**6.08 ± 0.78**
Cr other sectors	**0.78 ± 0.90**	**3.66 ± 1.78**	**2.50 ± 1.77**
Cu MWI plume	2.58 ± 0.20	27.22 ± 1.06	17.94 ± 1.23
Cu other sectors	2.71 ± 1.06	50.3 6 ± 17.57	16.74 ± 6.63
Pb MWI plume	3.50 ± 0.39	48.09 ± 1.57	22.57 ± 1.25
Pb other sectors	5.70 ± 2.85	70.1 4 ± 31.76	19.90 ± 6.46
Mn MWI plume	4.20 ± 1.19	9.72 ± 0.26	8.56 ± 0.72
Mn other sectors	5.19 ± 3.89	11.25 ± 2.75	9.67 ± 1.97
Ni MWI plume	0.60 ± 0.42	**14.26 ± 2.33**	8.34 ± 2.81
Ni other sectors	0.45 ± 0.42	**4.59 ± 5.83**	3.84 ± 6.33
V MWI plume	0.86 ± 1.13	2.44 ± 0.10	5.27 ± 1.13
V other sectors	0.91 ± 1.25	1.87 ± 1.22	2.45 ± 3.15

Table 10
Value for the ratios representative of MWI emissions reported in this study and in the literature.

	This study	Nielsen et al. (2010)	EMEP-CORINAIR (2006) Morselli et al. (2002)	Hu et al. (2003)	Mamuro et al. (1980)
Cu/Pb	0.83 [0.67–0.99]	0.24 [0.17–0.32]	(0.89 [0.39–0.84])·10^{-3}	–	–
Cd/Pb	0.08 [0.06–0.10]	0.08 [0.07–0.09]	0.03[0.03–0.03]	0.05	0.03
Cd/Cu	0.14 [0.12–0.170]	0.34 [0.28–0.41]	36.56 [83.61–17.66]	0.21	–
Cr/Pb	0.564 [0.38–0.75]	0.28 [0.25–0.32]	(0.02 [0.01–0.04])·10^{-1}	–	–

comprised only a short snapshot of the MWI output throughout the study period (daily emissions on quarterly basis). Metals emissions from the MWI might change weekly, seasonally and/or on yearly basis. However the standard deviation in the four metal ratios was small meaning that these could be used with confidence as MWI tracers.

The ambient dataset available for this study ranged from 2 to 5.5 years depending on the MWI (Table 1). It comprised weekly samples of heavy metals and hourly meteorological information. Wind direction dependent emission ratios have been used successfully elsewhere in receptor analysis; for instance Johnson et al. (2014) recently used V, Ni, sulphur and black carbon ratios to examine the influence of shipping emissions on ambient air pollution in Brisbane, Australia. Although daily variations of the source cannot be observed in the weekly samples, analysis on a test dataset from the Harwell – Didcot Power Station showed that weekly mean concentrations combined with hourly meteorological data can accurately detect a point source and track temporal changes under stable meteorological conditions. Following results from the Harwell – Didcot Power Station test, the analysis of UK MWIs focused on stable meteorological conditions. These were met between 51–68% of the time when the wind blew from the direction of the MWIs.

The fingerprint metal ratios from MWI stack emissions were found to be very different to those in ambient rural environments and those close to traffic. Particulate metals are a primary emission from MWI (Table 2) and bag-filtered stack emissions from MWIs do not contain a significant amount of particulates greater than 10 μm diameter (Buonanno et al., 2009; Ashworth et al., 2013). Over the maximum 10 km distance considered in the study the different particulate metals should therefore be subject to the same rates of dispersion and deposition. Although concentrations of particulate metals would be expected to decrease with distance from the stack, the emissions ratios will be conserved in the MWI plume.

Detecting stack emissions using ratios in ambient data is most likely to be successful if the stack is the only source of the tracer species. The presence of other sources emitting the same species at different rates might change the ratios in ambient data making difficult to isolate sources. Some studies have used the ratio of heavy metals (e.g. Cd) related to Pb to detect the influence of MWI emissions in urban ambient air (e.g. Sakata et al., 2000). However, Pb emissions in Europe mainly come from area sources such as traffic (Pacyna et al., 2007, 2009; Noble et al., 2008) while Cd is emitted primarily from point sources (e.g. waste incinerators). The dissimilar distribution of emissions of Cd and Pb would represent a challenge for the detection of MWI emissions in ambient air as the emissions from other sources would modify the ratio measured at the measurement site. Ambient ratios different from the rural and traffic values might indicate the presence of other sources emitting metals to the atmosphere (e.g. MWIs). Most of the MWI in the UK are located in heavily industrialized areas and these might also modify the ambient metal ratios. In order to overcome this type of confounding behaviour, we used four tracer ratios to identify emissions from MWIs. Our technique identified that traffic was the main source of metals in central London demonstrating its specificity. Despite three of the four ratios used to fingerprint MWI emissions being related (Cd/Pb, Cd/Cu and Cu/Pb) the combination provided specific source information. For example, at the end of the time series shown in Fig. 4 ambient values of Cu/Pb were within the MWI emissions value although Cd/Cu and Cd/Pb values clearly indicated the dominance of traffic emissions.

In summary we did not detect incinerator source profiles in ambient particulate matter metal concentrations around four UK MWIs. However, MWI emissions might still influence ground-level concentrations but the location of the sampling sites did not detect them. Despite the ambient sampling locations were not ideally placed to detect the influence of the MWIs (e.g. not downwind in the prevalent wind direction, near other metals emitting industrial sources, etc.) and the time resolution of measurements were only weekly samples, we successfully identified emissions from MWI for two installations in UK. Metal ratios consistent with MWI emissions were found in ambient measurements within 10 km of the Stockton MWI for about 5.4% of the time when the wind blew from the incinerator under stable conditions. The Wolverhampton MWI was similarly detected at two ambient metals sites, about 2.6% and 0.5% of the time when the wind blew from the incinerator under stable conditions. This was 0.2% of the total study period at Stockton and a maximum of 0.1% of the study period at Wolverhampton. Stockton-on-Tees and Wolverhampton are the second and third largest UK MWI in terms of daily PM emissions (Table 1), which might explain their detection in the study. Using metal tracers we estimated a maximum ambient PM from these two MWIs between 0.03 and 0.12 μg m^{-3} at our receptor sites. These concentration estimates were one to two orders of magnitude larger than the dispersion-modelled mean PM concentrations which were between 10^{-4} and $2 \cdot 10^{-3}$ μg m^{-3} at the metals sites. It must be remembered that our tracer method assumed that all Cd during plume grounding arose from the MWI which would lead to an overestimate of the ambient contribution. Importantly, however, both the emission ratio and dispersion modelled estimates were very low compared to background levels. Annual PM_{10} ambient levels ranged from 20 to 31 μg m^{-3} at urban background and roadside sites between 2003 and 2010 (DEFRA, 2014); 2–3 (compared to emission ratio estimates) and 3–4 (ADMS) orders of magnitude larger. It is not feasible to measure increments of this order of magnitude above background PM values using state-of-the-art instruments. For all the metals sites where MWI emissions were detected, higher of Cr concentrations were detected during the grounding periods compared with other wind sectors; Ni concentrations were also higher at 95% confidence interval for one metals site. This is consistent with the relative abundance of these metals in MWI emissions.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.atmosenv.2015.05.002.

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Long term plant biomonitoring in the vicinity of waste incinerators in The Netherlands

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HIGHLIGHTS

- Burning of waste can result in the emission of potentially toxic compounds.
- Biomonitoring can be used to monitor the impact of emissions on agricultural crops.
- Heavy metals, PAHs and dioxins/PCBs levels were similar to background levels.
- The fluoride standard for cattle feed was sometimes exceeded in grass samples.
- The results have contributed to a better relationship between stakeholders.

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ABSTRACT

Since the mid-nineties new waste incineration plants have come into operation in the Netherlands. Burning of waste can result in the emission of potentially toxic compounds. Although the incineration plants must comply with strict conditions concerning emission control, public concern on the possible impact on human health and the environment still exists. Multiple year (2004–2013) biomonitoring programs were set up around three waste incinerators for early detection of possible effects of stack emissions on the quality of crops and agricultural products. The results showed that the emissions did not affect the quality of crops and cow milk. Concentrations of heavy metals, PAHs and dioxins/PCBs were generally similar to background levels and did not exceed standards for maximum allowable concentrations in foodstuffs (e.g. vegetables and cow milk). Some exceedances of the fluoride standard for cattle feed were found almost every year in the maximum deposition areas of two incinerators. Biomonitoring with leafy vegetables can be used to monitor the real impact of these emissions on agricultural crops and to communicate with all stakeholders.

1. Introduction

Since the mid-nineties new waste incineration plants have come into operation in various agricultural regions in the Netherlands to accommodate the increasing amount of wastes produced. Burning of municipal solid waste can result in the emission of potentially toxic compounds including heavy metals and organics such as dioxins and polychlorinated aromatic hydrocarbons (Hutton et al., 1988; Bache et al., 1991; Schuhmacher et al., 1998; Loppi et al., 2000). Emissions and aerial dispersion of these compounds depend on waste composition, design of the waste incineration plant, operating conditions during combustion, emission control, stack height and prevailing weather conditions (Bache et al., 1991). The incineration plants in this study comply with strict conditions concerning emission control, and state-of-the-art technologies are used to remove gaseous components and fly ash. However, there was, and still is great deal of public concern about the possible impact of the emissions on human health, well-being and the environment. These include serious concerns about the possible effects on the quality of their crops grown in the direct vicinity of incineration plants. In order to meet these concerns, biomonitoring programs were set up in the direct vicinity of incineration plants to detect possible effects on agricultural crops and products. Furthermore, arrangements were made for financial compensation should such effects occur.

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Biomonitoring with plants is a technique to indicate effects of ambient phytotoxic compounds (Mannig and Feder, 1980; Tonneijck et al., 2002; De Temmerman et al., 2004). Biomonitoring is performed with bioindicator and accumulator plants. Bioindicators are sensitive plant species showing visible symptoms such as necrosis, chlorosis, abortion of flowers or fruits, or growth reduction. Accumulators are generally less sensitive than bioindicators, but accumulate gases and particles in or onto their leaves without showing visible effects, which can be measured and analysed. Gases are usually taken up into the leaves and particulates are accumulated on the leaf surface, while lipophilic organic substances are primarily accumulated in the waxy layers of plant tissues (Falla et al., 2000; Garrec and Van Haluwyn, 2002). Biomonitoring can be used to monitor spatial and temporal distributions of environmental effects around emitters of air pollutants (Klumpp et al., 2002; Ernst, 2003; Weiss et al., 2003), or can be used as an early warning system (Keddy, 1991).

This study describes the results of multiple year (2004–2013) monitoring programs around three waste incinerators in The Netherlands. The aim of these programs was the early detection of possible effects of stack emissions on the quality of crops and agricultural products. These programs focus on components mainly emitted through the flue gases: the heavy metals cadmium (Cd) and mercury (Hg), polycyclic aromatic hydrocarbons (PAHs), dioxins (PCDD/PCDF) or dioxin-like polychlorinated biphenyls (PCBs) and fluorides (HF).

Depending on the chemical composition and amount of the collected waste, various heavy metals are emitted during the incineration process (Morselli et al., 1993). Metals like Cd and Hg are not fully retained in the slag or captured in the electrostatic filter, but are emitted as gas with other flue gases (Morselli et al., 1993). Therefore Cd and Hg are relevant components to monitor around waste incinerators. Gaseous Hg is spread over large distances, and above-ground plant organs absorb gaseous Hg (Stoop et al., 1992). When fume temperatures decrease, Cd is adsorbed by suspended particles and spread through the air and can be taken up by plants via the stomata. Cd is very mobile and can be easily transported throughout the plant, and eventually be stored in different plant parts such as roots, stems and seeds (Stoop and Rennen, 1991).

During incineration, polycyclic aromatic hydrocarbons (PAHs) are released by the incomplete combustion of organic materials in the waste stream. The ratio between the various hydrocarbons depends largely on the conditions during the combustion process. Especially high molecular PAHs have a wide range of toxicological effects on human health, including cancer. Plants can take up PAHs from the air, which to a very limited extent, can then be transported to other plant organs. Elevated levels in plant shoots are generally the result of leaf absorption (Debus et al., 1989; Franzaring, 1995). The major uptake routes are active uptake of gaseous PAHs through the stomata and the passive diffusion through the cuticle. Plant species with a high lipid content, wide, curled leaves and a surface structure favourable for uptake (thick wax layer) accumulate PAHs relatively easily. An example of such a plant is kale, in Germany frequently used as monitoring plant in routine environmental research (Radermacher and Rudolph, 1994; VDI, 1999).

Dioxins have no technical application and are therefore not deliberately manufactured. Dioxins and dioxin-like PCBs originate from a variety of combustion processes, including waste incineration. The residence time of individual components in the air depends on their physical form. Depending on material properties and temperature, dioxins and dioxin-like PCBs occur in the gas phase or are adsorbed to airborne particles. In combination with the meteorological conditions this determines the deposition area (Liem et al., 1993). Dioxins have the ability to accumulate in body fat. Livestock that consume contaminated feed, accumulate dioxins in fatty tissue and milk fat which becomes part of the food chain. Fürst et al. (1992) estimated that up to 90% of human exposure to dioxins and dioxin-like PCBs results from the consumption of food, mainly animal origin. The other 10% comes from inhalation and ingestion of ambient particles. Exposure to dioxins and dioxin-like PCBs can result in a wide range of toxicological effects on human health, including cancer and disturbance of the reproductive and immune system (SANCO, 2001).

In general fluoride levels in grass follow a seasonal pattern, with higher levels in autumn and winter and lower levels in the summer. Dilution of accumulated fluoride in plant tissue with higher growth rates and surface roughness are important factors for this seasonal fluctuation (Van der Eerden, 1991). Adverse fluoride effects can occur in livestock (fluorosis: damage to bones and teeth) by consuming plants with accumulated fluorides. Humans are less sensitive to fluorides; effects on human health as a result of ambient fluoride are negligible (Mennen et al., 2010) and fluoride is even intentionally added to drinking water and toothpaste.

The aim of these programs was to assess the effects of potentially toxic components on the quality of crops and agricultural products in the vicinity of waste incinerators. In addition, a qualitative assessment was made of the values of these biomonitoring programs for farmers, stakeholders and waste treatment companies.

2. Methodology

A biomonitoring program was set up around waste incineration plants nearby Alkmaar (Incinerator 1; 52°36′35.04″N/4°45′44.57″E) and Wijster (Incinerator 2, 52°47′25.62″N/6°30′49.42″E), each with a capacity of approximately 800–1.000 kiloton annual and a stack height of 80 m. A similar program was started in 2010 around the incinerator nearby Harlingen (Incinerator 3, 53°11′31.11″N/5°25′47.41″E) with a capacity of approximately 250 kiloton annual and a stack height of 40 m. The three incinerators were chosen for the state-of-the-art technologies they use. Together they process about 30% of the total annually waste incinerated with energy recovery in The Netherlands (Eurostat, 2010). A network of five sampling points was set up in the agricultural areas around each incinerator. The program was performed in the same way at each location. A general dispersion model (STACKS) was used to predict the distance where the plume would reach the ground under the prevailing wind direction (southwest). Based on these calculations the locations around Incinerator 1 and 2 were placed at a distance of 3–4 km and around Incinerator 3 at 1.5–2 km because of the lower stack height. In each network one location was located where the deposition was considered maximal, northeast of the installations (maximum deposition area). Three other locations were situated in opposite wind directions (SE, SW and NW). A fifth location was situated at a larger distance (10–12 km) outside the immediate influence of the stack gases in order to monitor the regional background concentrations (reference location). Each sampling location was first provided with an anti-rooting plastic to suppress weed growth and prevent contamination of the plants with soil particulates from the immediate vicinity. A 1 m high fence and windscreen was set up to protect the plants against rabbit herbivory and wind damage.

Spinach (*Spinacia oleracea* L.) and kale (*Brassica oleracea* L.) were used as accumulator crops because of their high growth rate, large leaf area and growing season. Both species have been commonly used as accumulator crops (Franzaring, 1995; De Temmerman and Hoenig, 2004). Depending on the time of year spinach and kale were used to monitor the accumulation of Cd, Hg and PAHs. At each sampling point plants were cultivated in 50 dm^3 containers

with standard soil ('Lentse' soil No. 3, Horticoop, Bleiswijk, NL) with an automatic water supply (adopted from Posthumus, 1982). Spinach was harvested every four weeks in five consecutive exposure periods between April and August (Week 18, 22, 26, 30 and 34). Kale leaves were harvested at 8 week intervals during autumn and winter (Week 42, 50 and 6 of the next calendar year). Per sampling site all available leaf material was harvested, thoroughly mixed and dried at 40 °C for 48 h and subsequently ground to 1 mm and stored in plastic sampling containers. After extraction of the elements with concentrated nitric acid in a microwave concentrations of Cd and Hg were determined with Inductively coupled plasma mass spectrometry (ICP-MS), adopted from NEN-EN-ISO 17294 (2004). Concentrations of PAHs (16 EPA) were determined with gas chromatography/low resolution mass spectrometry (GC-LRMS) after extraction of the samples with toluol for 8 h and clean-up of the extract on a silica-gel column (adopted from VDI 3874, 2006).

Cow milk was sampled at two dairy farms (4 samples per year) in the vicinity of each incinerator to determine the concentrations of dioxins (PCDD/PCDF) and dioxin-like PCBs. Farms were selected where the cattle had mainly grazed in the maximal deposition area, or acquired part of their forage from that location. Samples were taken in week 22 and 38. The potential exposure of grazing lifestock was the highest. A one litre milk sample was taken from the storage tank (4 °C) and stored in a glass container and kept in cool storage during transport to the laboratory. Each sample was centrifuged and the upper layer (dry cream) was collected in a beaker with anhydrous sodium sulphate. After adding 150 mL of pentane, the extract was collected by decanting over a filter in a funnel and collected. The extraction was repeated two times. The sample was spiked with 13C-labeled standards (dioxins, furans, non-ortho PCBs, mono-ortho and non-dioxin-like PCBs). After drying overnight at 40 °C, the fat was weighted and cleaned using a Powerprep system. From the 210 different dioxin congeners, identification and quantification of the 17 most relevant dioxins and 12 dioxin-like PCBs was performed with gas chromatography/high resolution mass spectrometry (GC/HRMS), adopted from Tuinstra et al. (1994) and Hoogenboom et al. (2007).

The relative toxicity of dioxins and dioxin-like congeners was compared to the most toxic substance 2,3,7,8-TCDD (2,3,7,8-tetrachlorodibenzo-p-dioxin), the reference congener, and a toxic equivalency factor (TEF) was assigned. A toxic equivalency (TEQ) level was calculated by multiplying the actual level of each dioxin and dioxin-like compound by its corresponding TEF and then summing the results (Van den Berg et al., 2006).

Fluoride concentrations were measured in field-grown pasture grass. In the vicinity of each sampling point grass from a sufficiently large and homogeneous pasture was sampled at four-weeks intervals (13 samples per year). The grass was cut just above the ground at 16 points in a grid of 9 × 9 m. The grass was mixed and dried at 105 °C for 48 h and subsequently ground to 0.5 mm and stored in plastic sampling containers. Fluoride concentrations were determined by ashing 0.5 g of grass for 2 h at 520 °C. The ash was dissolved in NaOH and water. Fluoride was distilled from the solution with sulphuric acid at 170 °C, condensed and mixed with a colouring agent followed by colorimetric measurement at 620 nm (adopted from Weinstein et al., 1972; CBLB, 2011).

3. Results and discussion

3.1. Effects on the quality of crops and agricultural products

To prevent the influence of pollutant uptake from the soil, spinach and kale plants were cultivated in containers with unpolluted standard soil. Consequently, pollutants in above-ground plant organs are considered to be the result of aerial uptake. Indicative background levels of cadmium, mercury and PAHs in spinach and kale were calculated from the concentrations measured at all three reference locations over the last five years (moving five-year average and standard deviation). With these reference data, longer-term trends can be highlighted and the average levels measured in the direct surroundings of each incinerator can be distinguished from the background level for the Netherlands. The range in the moving five-year average indicates the variation in time and whether there is an upward or downward trend. The same approach was used for the background level of fluorides in grass, and included the results of three additional sampling points in non-polluted areas in The Netherlands. The background level of dioxins and dioxins-like PCBs was based on an inventory of the Dutch Institute of Food Safety over the period 2001–2010 (Schoss et al., in prep.).

3.1.1. Heavy metals

The multiple year results from the biomonitoring programs in this study showed that individual Cd levels in spinach and kale exhibited some variation between sampling points and throughout the season, including the reference sampling point. Part of the variation can be explained by differences in plant growth, diluting the accumulated metals. This variation explains the bandwidth of the background level. The annual mean Cd levels in spinach from 2004 to 2013 from the four sampling points in the direct vicinity of each incinerator were within the range of the background level (Fig. 1, left). The order of magnitude of the Cd levels was similar for the three incinerators, despite the differences in incineration capacity and emission flows. The capacity of incinerators 1 and 2 is three to four times higher than for incinerator 3. The 2010 average levels around incinerator 3 refer to the background level in that year because the incinerator was still under construction, and only became operational in the beginning of 2011. The higher levels in 2010 measured around incinerators 1 and 2 are therefore not likely the result of higher emissions, but higher background concentrations. The Cd concentrations in kale were lower than those in spinach, and were within the range of the background level (Fig. 1, right). Higher levels occurred around incinerator 1 and 2 only in 2006. They resulted from higher Cd levels at all sampling points including the reference points in autumn (October) and early winter (December). Thus, it is unlikely that the emission of the incinerators was responsible for these higher levels in kale. The maximum acceptable Cd level for leaf vegetables (200 μg kg^{-1} fresh weight) to protect public health (EC, 2008) was not exceeded.

Hg levels in both spinach and kale were relatively low throughout the whole monitoring period (Fig. 2), even though Hg is one of the few heavy metals mainly emitted as gaseous component by waste incinerators. Vapour phase Hg is mainly due to the incineration of discarded batteries (Bergström, 1986), and domestic solid waste in The Netherlands contains only 0.03% discarded batteries (Ministry of Infrastructure and Environment, 2013) due to a high degree of battery recycling. This could well be an explanation for the relatively low Hg levels found in spinach and kale. Furthermore gaseous mercury is distributed over a larger area than particle-bound cadmium. These findings are consistent with low Hg concentrations found in lichens around an incinerator in central Italy (Loppi et al., 2000). No maximum acceptable Hg level for leaf vegetables has been defined.

3.1.2. PAHs

Regarding heavy metals, individual PAH levels in spinach and kale were variable between sampling points and throughout the season. Variation in PAH levels also occurred at the reference sampling point, which explains the bandwidth of the background level. PAH levels in crops are correlated to seasonal fluctuations

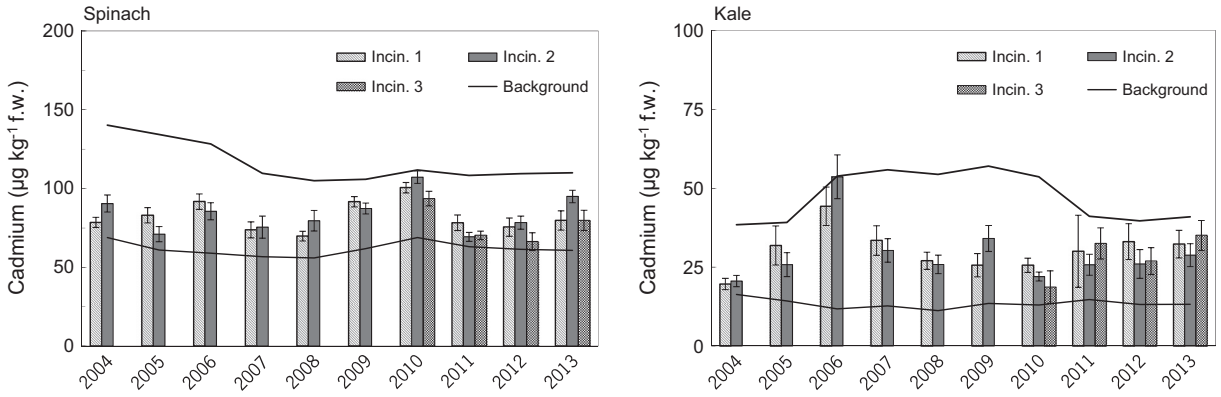

Fig. 1. Mean (±SE) cadmium levels (µg kg^{-1} f.w.) in spinach (left) and kale (right) exposed to ambient air in the vicinity of three waste incinerators from 2004 till 2013. The solid lines indicate the range for the background level. The maximum acceptable Cd concentration in leaf vegetables to protect public health is 200 µg kg^{-1} f.w. (EC, 2008).

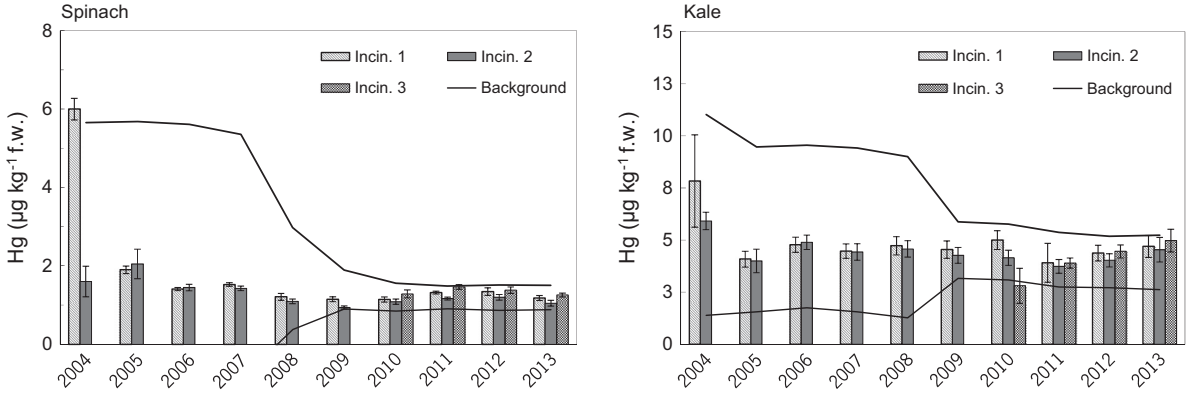

Fig. 2. Mean (±SE) Hg levels (µg kg^{-1} f.w.) in spinach (left) and kale (right) exposed to ambient air in the vicinity of three waste incinerators from 2004 till 2013. The solid lines indicate the range for the background level.

corresponding to the fluctuations in the local air temperature and particle concentration. During periods with lower temperatures PAH emissions increase due to heating and combustion processes. At the same time, the degradation of atmospheric PAHs is relatively low due to lower UV radiation levels in winter compared to the summer period. Decreasing temperatures also lead to increasing condensation of PAHs on airborne particles (Franzaring, 1995; Sucharova and Hola, 2014).

From 2004 to 2013 the PAH levels in spinach and kale, averaged annually for the four sampling points in the direct vicinity of each incinerator, were generally within the range of the background level (Fig. 3). In the summer of 2004 the PAH levels in spinach around incinerators 1 and 2 were higher than the upper limit of the background level. No unexpected deviations in the emission pattern were detected and higher levels occurred at all sampling points, including the two reference points. Therefore, it is unlikely that the emission of the incinerators was responsible for these higher levels. PAH levels in spinach around incinerator 3 were slightly lower than levels around incinerators 1 and 2, which could be due to its lower capacity. However, these differences in kale

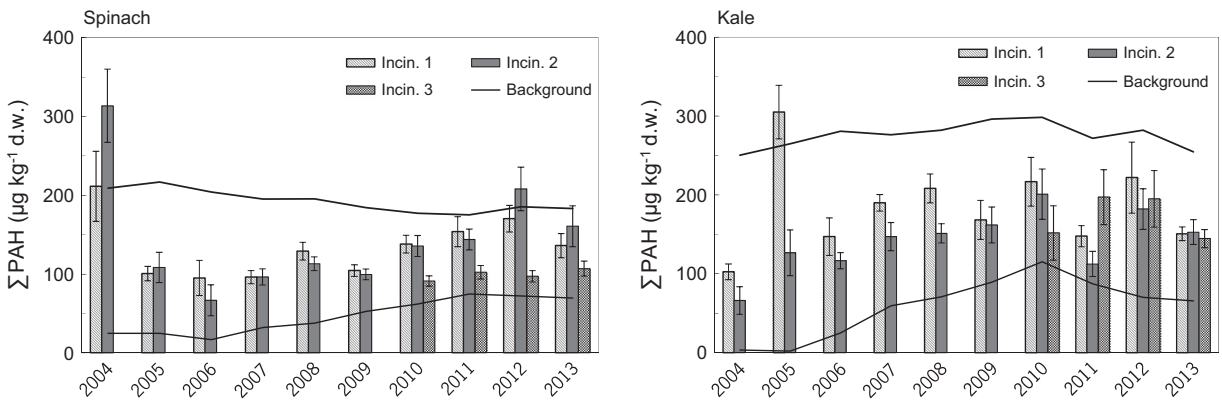

Fig. 3. Mean (±SE) PAH levels (µg kg^{-1} d.w.) in spinach (left) and kale (right) exposed to ambient air in the vicinity of three waste incinerators between 2001 and 2011. The solid lines indicates the range for the background level.

were not detected. PAH levels in kale around incinerator 1 in 2005 were above background levels and were also higher compared to incinerator 2 with a comparable capacity. The higher PAH levels in kale in the winter were the result of high levels of individual PAHs, benzo(a)anthracene, chrysene and benzo(b)fluoranthene. Other individual PAHs (13) were below the detection limit. There was no a statistically significant relationship between the higher PAH levels and the frequency of wind (hours) in the direction of the incinerators towards the sampling points.

3.1.3. Dioxins

The levels of dioxins (mean of 4 samples per year) in milk from the dairy farms in the immediate vicinity of the incinerators are comparable to the average background level for The Netherlands, i.e. 0.34 pg TEQ g^{-1} fat. The dioxin concentrations were relatively constant over time. The dioxin-like PCB concentrations showed a slight tendency to decrease over the years, which is in line with the results from the national survey from 2001 to 2010 (Schoss et al., in prep.). Results from a European survey showed that average dioxins and dioxin-like PCB levels in milk were about twice as high as the concentrations found in this study (EFSA, 2010). Dioxins and PCB levels do not appear to be related to emissions from the incinerators (Fig. 4). The levels of dioxins and the sum of dioxins and PCBs remained well below the maximum permissible level for milk and milk products of 2.5 and 5.5 pg TEQ g^{-1} fat respectively (EC, 2011). The results show that there is no potential risk with respect to the consumer quality of the examined milk.

3.1.4. Fluoride

Fluoride levels in pasture grass in the vicinity of the incinerators were generally at background levels, and followed the usual seasonal patterns. However, the annual average fluoride concentrations in grass in the immediate vicinity of incinerator 1 were higher than the background levels for several years (Fig. 5). Particularly in the winter period, levels were higher than expected based on the seasonal background level. The strictest standard for feed for young cattle (25 μg g^{-1} DW) set by the Dutch Health Council (Gezondheidsraad, 1990) was exceeded in most years (the European standard for feed for cattle, sheep and goats in lactation is 30 μg g^{-1} DW; EC, 2002). Fluoride levels in the summer were relatively low. In 2004–2007, 2009 and 2012 a significant relationship ($p < 0.05$) was found between the fluoride concentration and the frequency of wind (hours) from the incinerators towards the sampling points. Although a causal relationship was not proven and the correlation coefficients were low, a contribution of the emissions to the fluoride levels found could not be completely excluded. For these years, no increased concentrations were found in the continuous stack-measurements of HCl, a parameter with an emission pattern expected to be similar to that of fluorides. In

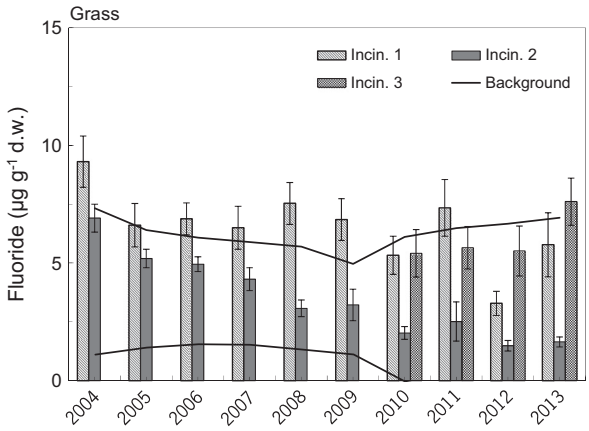

Fig. 5. Fluoride (μg g^{-1} d.w.) in pasture grass harvested in the vicinity of three waste incinerators. The maximum acceptable concentration in feed for young cattle is 25 μg g^{-1} dw (Gezondheidsraad, 1990).

2013, in the vicinity of incinerator 3 a similar pattern was found with higher fluoride levels in winter and several exceedences of the maximum standard for feed. From 2004 to 2013 the fluoride levels in grass around incinerator 2 were within the range of the background level, showing a decreasing trend. With regard to the risk to livestock, the fluoride levels in grass were of little significance.

Incinerators 1 and 3 are located relatively close to the coast. The contribution of sea salt to the fluoride concentration in the air could be a possible explanation of the higher fluoride levels. Sea water contains about 33 g l^{-1} of minerals, including 1.3 mg l^{-1} fluoride (CRC, 1989). The concentration of salt in the air in the coastal areas is approx. 5 μg m^{-3} (Hoogerbrugge et al., 2011). This means that the content of fluoride in the air as a result of the emission from the sea is about 0.0002 μg m^{-3}. This contribution is negligible compared to a background concentration of approx. 0.05 μg m^{-3} and no other mechanisms for fluoride emissions from seawater are known.

3.2. Stakeholders response to the monitoring programs

3.2.1. Farmers and residents living nearby the installations

Emissions from waste incineration plants contain components that are potentially toxic for humans, plants and animals. Major concerns often exist concerning the group of persistent organic compounds ('dioxins'). Therefore, plans for the construction of new installations meet resistance from local residents and environmental groups due to the fear of potential adverse health effects,

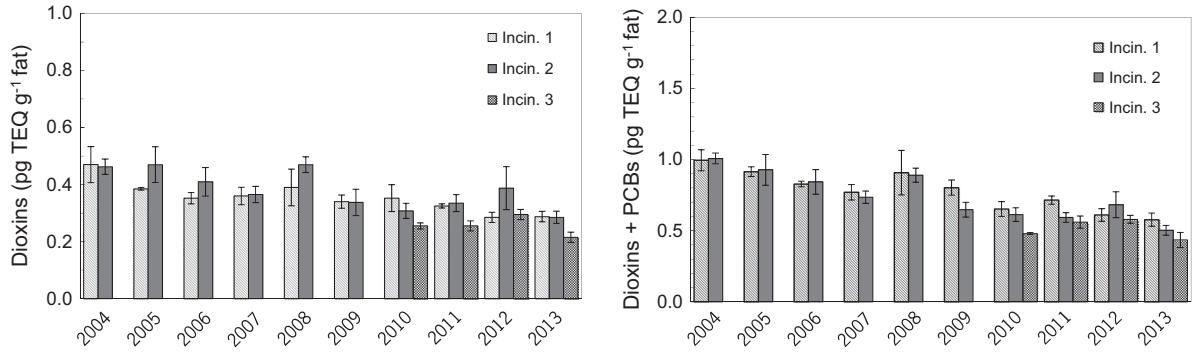

Fig. 4. Levels of dioxins (left) and dioxins + dioxins-like PCBs (right) in milk from dairy cattle in the vicinity of three waste incinerators (pg TEQ g^{-1} fat). The maximum acceptable concentration for dioxins in milk en dairy products is 2.5 pg TEQ g^{-1} fat. For dioxins + dioxins-like PCBs the standard is 5.5 pg TEQ g^{-1} fat (EC, 2011).

environmental contamination and the association of these facilities with odours, noise, visual intrusion, and the reduction in value of land and property. Major resistance also occurred in this case, when de plans for building these facilities became public. Whether the fear for health effects was justified cannot be answered with certainty. The results of various studies on the causal relationship between human health effects (reproduction, cancer, respiratory and heavy metals in the body) and exposure to emissions from incinerators are inconclusive (Hu and Shy, 2001; DEFRA, 2004). However, in general the (planned) presence of a incineration plant is perceived to be a potential risk by local residents (risk perception). Lima (1996) showed that risk perception can have an effect on the quality of life (health, environment, income) and mental well-being by introducing a dimension of danger in the residential environment. Furthermore, there are continuous reminders (noise, smell, smoke) of the closeness of the installation which requires extra effort (stress responses) to minimize the risks and adapt to this permanent threat (Lima, 2004).

Although it was not the primary aim of this research, the predominantly positive results (no effect) from the biomonitoring programs during the past decade have contributed to less uncertainty concerning health effects by neighbouring farmers and residents. The method of measuring ambient pollutants in plants grown at ground level, is probably closer to the real world of farmers and residents than the (legal mandatory) stack measurements, and therefore more convincing. The farmer's representation in the advisory committees of these programs also made it possible to discuss points of criticism and concerns. In the annual advisory committee meetings, an open communication existed on the program results and developments in the agricultural sector. In return, positive developments in the position and functioning of the waste incinerator took place. Confidence and trust (good-neighbour ship) were built up over time, resulting in present absence of farmer's resistance against the waste incineration installations.

3.2.2. Waste incinerating companies

Poor performance and incidents with (unintended) emissions of flue gases from mainly older incineration plants made waste incineration controversial in the past. Nowadays, most of the old incinerators are closed and new installations have come into operation. The companies involved are striving for the most sustainable waste processing with the best possible protection of the environment. Outside of formal permission or regulatory processes, a good relationship with the farmers and residents living nearby the installations have become increasingly important. Differences in perception between company and residents in the past often resulted in time and money consuming lawsuits and negative imaging in the press. In order to meet the public concerns, the biomonitoring programs were set up to detect possible effects on crops in the vicinity of the incineration plants indicating their potential effects on the environment. These programs and the proven absence of significant negative effects have contributed to achieving the objectives of sustainability and social responsibility of the companies involved.

3.3. Future developments

It is becoming increasingly more important to consumers and retail businesses that agricultural products are produced safely and hygienically. Due to this development, more and more farmers are working with food safety certification systems (GLOBALGAP). Certification aims to ensure the food safety during cultivation, harvest, transport and storage of crops for processing. Biomonitoring can contribute to the quality assurance of cultivated crops and products by demonstrating the absence of important ambient pollutants such as heavy metals, polycyclic aromatic hydrocarbons and dioxins.

Due to overcapacity, commercially-operated Dutch incinerators are importing municipal waste from other countries, mainly from the UK, Germany and Italy (approx. 300.000 ton in 2012) of which the exact composition is not always known. For the coming years it is expected that this import will further increase to 2–3 million tons per year, approx. 25–30% of the total incinerating capacity in The Netherlands (Rabo, 2012). By using state-of-the-art technologies and following good waste management practices, impacts on the environment can be minimized. However, from the point of view of the local community, this development could also become a strong argument for implementing new or continuing existing biomonitoring programs.

4. Conclusions

This study shows that leafy vegetables can be used for monitoring the impact of atmospheric deposition, which is consistent with previous research (Franzaring, 1995; VDI 3957/3, 2000; De Temmerman and Hoenig, 2004). The multiple year (2004–2013) results of monitoring in the vicinity of waste incinerators showed that the emissions did not affect the quality of crops and cow milk. Concentrations of heavy metals, PAHs and dioxins/PCBs were generally similar to background levels and did not exceed standards for maximum allowable concentrations in foodstuffs (e.g. vegetables and cow milk). The results also show that there is no potential risk with respect to human consumption quality of the investigated crops and products in the vicinity of the incinerators. Some exceedances of the fluoride standard for cattle feed were found almost every year in the maximum deposition area of the incinerators 1 and 3. A contribution of the emissions to these levels found could not be completely excluded. However, in absolute terms these levels are of minor significance with respect to the risk to livestock and have been accepted as such by the farmers.

Biomonitoring can be used to monitor the impact of ambient emissions from incinerator facilities on agricultural crops and products. The positive results (no effect) in combination with the open communication between stakeholders have also contributed to less uncertainty concerning potential health risks and have contributed to a better relationship between the farmers, residents and the companies involved. Future developments regarding increasing waste import from other countries is a strong argument for the implementation of new or the continuing of existing biomonitoring programs in the vicinity of incinerators.

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