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Review

The Impact of Industrial Fires in Urban Settings: Monitoring, Modelling, Health, and Environmental Justice Perspectives

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Abstract: Industrial fires at facilities including waste management sites, warehouses, factories, chemical works and fuel storage depots are relatively frequent occurrences. Often, these fires occur adjacent to urban communities and result in ground-level airborne pollutant concentrations that are well above guideline values. Land, water, livestock and crops may also be contaminated by the emissions and by firefighting activities. Moreover, impacted communities tend to have a higher proportion of minority ethnic populations, individuals with underlying health vulnerabilities, as well as those of lower socio-economic status. Nevertheless, this is an aspect of air quality that is under-researched, and so this review aims to highlight the public health hazards associated with industrial fires and the need for an effective, coordinated, public health response. We also review the range of monitoring techniques that have been utilised in such fires and highlight the role of dispersion modelling in predicting plume trajectories and in estimating population exposure. We recommend establishing 1-h guideline values for particulate matter to facilitate timely public health interventions and we highlight the need to review regulatory and technical controls for sites prone to fires, particularly in the waste sector.

Keywords: industrial fires; airborne pollution; public health hazard; particulate matter (PM); Monitoring techniques; dispersion modelling; emergency response; waste management; landfill; guideline values

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1. Introduction

Airborne pollution from industrial fires represents a significant but under-researched public health hazard. These are fires that occur at chemical works, factories, warehouses, fuel storage depots, landfills, recyclables storage sites, and other similar facilities [1]. Many industrial fires occurred in close proximity to residential areas, that may also have sensitive receptors such as schools, hospitals and care homes for the elderly [2,3]. And affected communities may have a disproportionate number of disadvantaged or vulnerable inhabitants [4]. For many incidents, the resultant ambient air pollution concentrations often exceed guideline values (GVs), especially for particulate matter (PM). For example, Griffiths, et al. [5] in an analysis of monitoring data from 23 industrial fires, found incident-average concentrations for PM_{10} (PM with an aerodynamic diameter <10 µm) of up to 1450 µg m^{-3}, compared to the WHO 24-h guideline of 50 µg m^{-3}, with some 15 min averaged concentrations in excess of 6500 µg m^{-3}. Modelling studies carried out on an open-cut coal mine in Australia predicted $PM_{2.5}$ (PM with an aerodynamic diameter <2.5 µm) concentrations as high as 3700 µg m^{-3} during the initial phase of the fire [6], compared to the WHO 24-h guideline value of 15 µg m^{-3}. And for a tyre fire in the UK town of Mexborough, a modelling study by Griffiths [7] predicted that as many as 7800 residents may have been exposed to 24-h averaged PM_{10} concentrations that exceeded the US EPA AQI category of 'Hazardous' (>425 µg m^{-3}). Exceedances of GVs have also been documented for other harmful pollutants such as benzene [4], nitrogen dioxide (NO_2), hydrogen cyanide (HCN), Hydrogen bromide (HBr) and hydrogen chloride (HCl) [1].

The frequency of industrial fires adds to the public health concern. For example in Serbia there were 1715 landfill fires in 2021 alone, affecting 42 million acres of land, with many of the fires involving illegally dumped waste [8]. Data from Sweden shows that there were 20.4 waste-fires per year over the period 2012 to 2018, with the frequency and intensity increasing with time [3]. In Istanbul, Turkey, for the period 2015 to 2020 there were 844 factory fires, whilst in the United States there were 37,910 fires at industrial and production facilities over the period 2011 to 2015 [2]. For the UK, where there is the national Air Quality in Major Incidents (AQinMI), response service, there were 74 fires that were considered to be ‘major incidents’ over the period 2009 to 2018, though the total number of fires in industrial and commercial premises is much higher [1,5]; for example, there were approximately 400 fires that the UK Health Security Agency were involved with in 2021 alone [9]. There is a concern that because some fires at waste management sites are secondary to adjacent wildfires that a warming climate may increase their frequency in the future [3]. Griffiths, et al. [5] note a seasonal trend in fire incidents, with summer months having a greater number of fires.

One of the aims of this review is to highlight the issue of air pollution hazards from industrial fires, facilitating a wider and better understanding of the public health implications and the need for appropriate planning for such events. We examine the range of different fire sources and their main emissions, how monitoring and modelling of such fires is carried out, the associated health effects, the suitability of current guideline values, the implications for justice and the need for further research. Our viewpoint is informed from previous involvement in the UK AQinMI service, as lead for one of the monitoring teams and in overall incident management. We have also recently published analyses of the monitoring data that was collected from the AQinMI service over the period 2009 to 2018 [1,5,7].

2. Types of Industrial Fires and Associated Pollutants

In this section we discuss the literature concerning the main combustion products emitted during uncontrolled fires at a range of industrial facilities. We look at ground-level plume concentrations that have been monitored during such fires, as well as pollutant concentrations in other environmental media, for example as a result of contamination by ash and/or firefighting activities.

Whilst all industrial facilities have a potential to catch fire and release harmful pollutants into the atmosphere, it has been shown that certain industrial categories, such as waste management, are more frequently involved [1]. This is likely due to poor on-site management such as the lack of separation of materials at waste management sites. Moreover, lower-risk facilities such as waste transfer sites are often regulated under less stringent regulations compared to high-risk chemical processes, which are very tightly regulated because of historical incidents (e.g., Seveso [10]). In addition, waste management is an industrial sector that can give rise to unregulated (illegal) accumulations of waste that are consequently more prone to fires [8]. Griffiths, et al. [1] have produced a classification of the industrial categories involved in UK fires (Figure 1) which shows that waste fires (tyres accumulations, mixed recyclables, residual recyclables and electrical waste) accounted for 87.1% of all significant major incident fires, with just one single chemical incident (pesticide manufacturer).

The materials burning, together with temperature and oxygen supply, will determine the mix of air pollutants and their emission rates from an industrial fire, whilst meteorological conditions will determine the extent of interaction between the plume and populations in adjacent urban areas. This said, the identification and quantification of airborne pollutant components will be dependent on the choice of monitoring technology employed. For example the Gasmet FTIR approach used to measure volatile organic and inorganic substances (VOCs and VICs respectively) in the UK AQinMI scheme has only 24 pre-set species that can be monitored, whereas mass spectral detection techniques will not be restricted in this way [1]. Thus, Brilli, et al. [11] detected 132 separate VOCs in the smoke from biomass burning using proton transfer reaction time-of-flight mass spectromic determination, and Koss, et al. [12] reported a similar numbers of components.

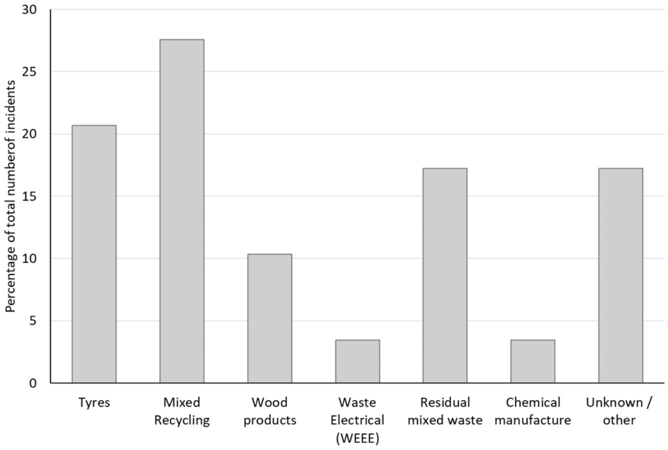

Figure 1. Relative frequency of different sources of major incident fires in the UK for the period 2009 to 2018 [1].

In a major systematic review of airborne pollutants released during the burning of various types of material, Lemieux, et al. [13] based their search on the 189 hazardous air pollutants (HAPs) mentioned in part III of the US Clean Air Act Amendments 1990. The data originated from both uncontrolled and controlled burning situations. The findings, summarised in Table 1, show that benzene, toluene, formaldehyde, acetaldehyde and acetone are major components of the combustion products for all categories, in terms of emission rate, as will be particulate matter (PM), though this is not reported. Semi-volatile compounds and those components that might adsorb onto PM, such as polycyclic aromatic hydrocarbons (PAHs) and polychlorinated dibenzodioxins/furans (PCDD/F) have also been quantified for many of the categories.

Table 1. Emission rates of key airborne pollutants during the burning of materials that might typically be components of industrial fires. Summarised from Lemieux, et al. [13].

	Emission rates (mg kg-1 unless otherwise stated) for different categories of substance				
Material burning	**VOCs**	**sVOCS**	**Carbonyls**	**PAHs**	**PCDDs/Fs**
Fuel oil	Benzene, 1022; toluene,42; ethyltoluenes,22; xylenes, 25; 1,2,4-trimethylbenzene, 32.	Not listed	Formaldehyde, 303; acetaldehyde, 63; acrolein, 39; acetone, 35; benzaldehyde, 104.	Naphthalene, 162; acenaphthylene, 99; fluoranthene, 20; 1-methylfluorene, 26; anthracene, 15.	HpCDD, 7.07 x 10^{-5}; OCDD, 1.34 x 10^{-4}; TCDF, 2.05 x 10^{-4}; HxCDF, 1.86 x 10^{-5} (data for crude oil).
Household waste	Benzene 980; styrene 527 ; toluene 372 ; ethylbenzene 182 ; chloromethane 163.	Phenol 113 ; 3- or 4-cresol 44.1 ; 2-cresol 24.6 ; bis(2-ethylhexyl) phthalate 23.8 ; isophorone 9.3.	Formaldehyde 444 ; acetaldehyde 428 ; acetone 254 ; benzaldehyde 152 ; propionaldehyde 113.	Naphthalene 11.4 ; phenanthrene 5.3 ; fluorine 3.0 ; fluoranthene 2.8 ; chrysene 1.8.	Total PCDD/Fs: 5.8 x 10^{-3}; TEQ‡ PCDD/Fs: 7.7 x 10^{-5};
Burning of scrap tyres	Benzene, 2180; toluene, 1368; styrene, 653; ethylbenzene, 378; limonene, 460.	Phenol, 533; 1-methylnaphthalene, 279; 2-methylnaphthalene, 390; bis(2-ethylhexyl)phthalate, 23.8; isophorone, 9.3.	Formaldehyde, 444; acetaldehyde, 428; acetone, 254; benzaldehyde, 152; propionaldehyde, 113.	Acenaphthene, 1368; naphthalene, 651; phenanthrene, 245; fluoranthene, 398; pyrene, 93.	Not listed

Automobile shredder residue	Toluene, 10690; benzene, 9584; styrene, 6528; ethylbenzene, 4964; chlorobenzene, 1718.	Bis(2-ethylhexyl) phthalate, 2058; benzenebutanenitrile, 3340; phenol, 990; benzaldehyde, 1690; 1,2-dichlorobenzene, 110.	Not listed	Naphthylene, 883.3; acenaphthylene, 150.0; fluorene, 38.0; phenanthrene, 231.3; anthracene, 35.7.	TCDD; PeCDF, 1.40; TCDF, 1.80; HxCDF, 0.40; PeCDD, 0.30.
Wood burning (tropical forest)	Benzene, 400; toluene, 250; acetonitrile, 180; methyl chloride, 100; xylenes, 60.	Furan, 480; 2-methylfuran, 170; 3-furfural, 370; 2,5-dimethylfuran, 30; 3-methylfuran, 3.	Formaldehyde, 1400; methanol, 2000; acetaldehyde, 650; acetone, 620; 2,3-butanedione, 920; acrolein, 180.	Total PAHs, 25.0.	Total PCDDs/Fs, 0.0067)

‡ Toxic equivalency.

A further view of airborne pollutants that are likely to be emitted during the burning of different source materials is given in Table 2, which lists specific incidents (or groups of incidents according to combustion material type) and the associated concentration data. Also, included in the first four entries are the results of a review of emissions by source type, compiled by Wakefield [14], which identifies the most significant hazardous components, and identifies those likely to be in breach of guideline values. The incidents listed in Table 2 include fires at chemical works, landfill sites and a coal mine. Often, the monitoring reported in Table 2 involves a limited range of compounds, emphasising the importance of the choice of monitoring technology, as discussed further in Section 3. Moreover, the recorded concentrations will be dependent on the monitoring locations relative to the fire plume: they may be fixed existing ambient stations, or mobile monitoring points chosen to assess public exposure. An example of the latter is the UK‘s AQinMI scheme which is a national monitoring service comprising standardised equipment, and involving the deployment of two mobile monitoring teams to the fires [1,5]. The results of this monitoring, averaged by source material type, is shown in entries 5 to 9 of Table 2; it also allowed the ‘fingerprinting’ of emissions for different source materials so as to predict the likely substances that will be emitted in such fires as shown in Figure 2. Thus, xylenes were a significant part of the emissions from tyre fires whereas formaldehyde and NO_2 are major components of fires involving wood. These findings were broadly in line with the characterisations of plume components reviewed by Lemieux, et al. [13] in Table 1.

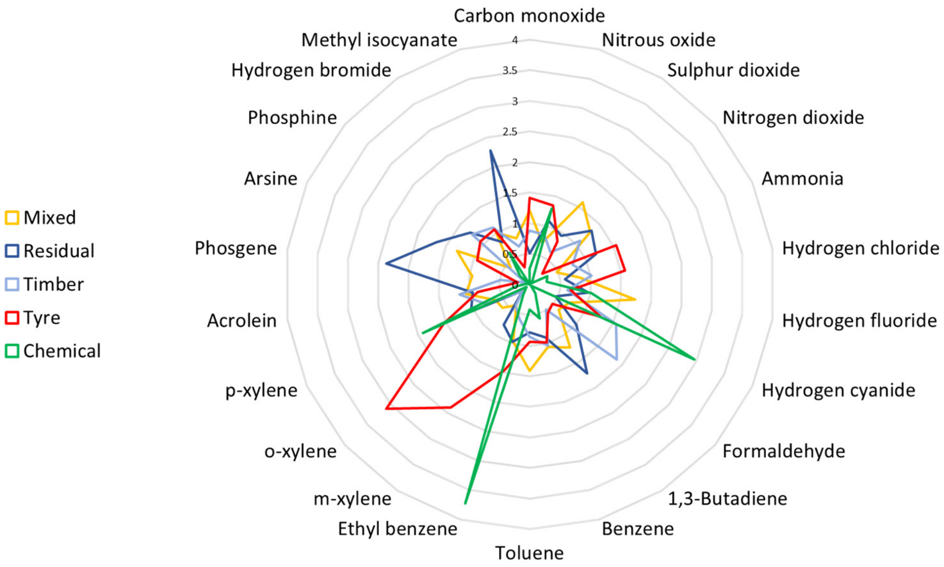

Figure 2. Polar plot showing relative emissions for a range of different industrial fire source-materials. Replotted from Griffiths, et al. [1].

Table 2. Measured concentrations of the main organic, inorganic and particulate air pollutants released during a range of industrial fires.

Incident or fire type	**Details**	**Airborne pollutants and concentrations (if known)**	**Reference**
Burning of polymeric materials	Qualitative data from a review of such fires	CO, HCN, HCl/HBr/HF, NO_x, SO_2, **organic irritants (acrolein/formaldehyde)**, inorganic irritants (phosgene/ammonia), PAHs, **PCDD/Fs**, **PM (bold indicates likelihood of high concentrations relative to standards).**	[14]
Burning of wood materials	Qualitative data from a review of such fires	NO_x, acrolein/formaldehyde, PAHs, PCDD/Fs, PM	[14]
Burning of rubber tyres	Qualitative data from a review of such fires	CO, HCN, HCl/HBr/HF, NO_x, **SO_2**, P_2O_5, **organic irritants (acrolein/formaldehyde)**, inorganic irritants (phosgene/ammonia), PAHs, **PCDD/Fs, PM (bold indicates likelihood of high concentrations relative to standards).**	[14]
Burning of oil and petrol	Qualitative data from a review of such fires	SO_2, **organic irritants (acrolein/formaldehyde)**, PAHs, PCDD/Fs, **PM (bold indicates likelihood of high concentrations relative to standards).**	[14]
Tyre fires	Based on ambient monitoring carried out as part of the UK's AQinMI service	Incident-mean concentrations (ppb for gases and $\mu g\ m^{-3}$ for PM) with the maximum 1 min concentration in parentheses: CO, 0.58 (3.01); HCN, 0.09 (0.48); ammonia, 0.22 (1.07); HBr, 0.53 (2.76); HCl, 0.05 (0.47); HF, 0.02 (0.28); NO, 0.23 (0.42); NO_2, 0.03 (0.74); phosgene, 0.01 (0.24); SO_2, 0.08 (0.7); benzene, 0.16 (1.13); 1,3-butadiene, 0.03 (1.45); ethyl benzene, 0.08 (1.31); formaldehyde, 0.03 (0.2); methyl isocyanate, 0.02 (0.22); m-xylene, 0.02 (0.62); o-xylene, 0.1 (1.5); p-xylene, 0.09 (0.72); toluene, 0.16 (3.74); arsine, 0.02 (0.12); PM_{10}, 115 (6527); $PM_{2.5}$, 27.8 (652)	[1,5]‡
Fires involving mixed recycling	Based on ambient monitoring carried out as part of the UK's AQinMI service	Incident-mean concentrations (ppb for gases and $\mu g\ m^{-3}$ for PM) with the maximum 1 min concentration in parentheses: CO, 0.78 (10.7); HCN, 0.17 (46.1); ammonia, 0.14 (4.74); HBr, 0.66 (5.66); HCl, 0.07 (66.2); HF, 0.08 (5.56); NO, 0.14 (2.32); NO_2, 0.11 (0.77); phosgene, 0.17 (0.85); SO_2, 0.68 (2.52); benzene, 0.2 (15.5); 1,3-butadiene, 0.49 (3.15); ethyl benzene, 0.1 (8.54); formaldehyde, 0.12 (0.88); methyl isocyanate, 0.01 (0.46); o-xylene, 0.01 (0.7); p-xylene, 0.23 (1.48); toluene, 0.36 (3.76); arsine, 0.01 (0.79); PM_{10}, 83 (6527); $PM_{2.5}$, 24.4 (652)	[1,5]‡
Fires involving timber	Based on ambient monitoring carried out as part of the UK's AQinMI service	Incident-mean concentrations (ppb for gases and $\mu g\ m^{-3}$ for PM) with the maximum 1 min concentration in parentheses: CO, 0.48 (12.3); HCN, 0.23 (24.7); ammonia, 0.16 (11); HBr, 0.7 (44.3); HCl, 0.06 (11.2); HF, 0.07 (7.81); NO, 0.07 (2.21); NO_2, 0.5 (147); phosgene, 0.25 (17.7); SO_2, 0.09 (9.51); benzene, 0.24 (6.93); 1,3-butadiene, 0.16 (23.9); ethyl benzene, 0.08 (6.81); formaldehyde, 0.44 (23.5); methyl isocyanate, 0.01 (0.35); m-xylene, 0 (0.98); o-xylene, 0.01 (1.28); p-xylene, 0.2 (6.27); toluene, 0.34 (23.2); arsine, 0 (1.76); PM_{10}, 28 (848); $PM_{2.5}$, 12.6 (306)	[1,5]‡
Fires involving WEEE materials	Based on ambient monitoring carried out as part of the UK's AQinMI service	Incident-mean concentrations (ppb for gases and $\mu g\ m^{-3}$ for PM) with the maximum 1 min concentration in parentheses: CO, 0.41 (2.58); HCN, 0.05 (0.49); ammonia, 0.03 (0.18); HBr, 0.52 (2.34); HCl, 0.02 (0.22); HF, 0.07 (0.31); NO, 0.04 (0.32); NO_2, 0.04 (0.32); phosgene, 0.06 (0.39); SO_2, 0.29 (0.82); benzene, 0.38 (1.34); 1,3-butadiene, 0.1 (0.73); ethyl benzene, 0.0 (0.07); formaldehyde, 0.02 (0.25); methyl isocyanate, 0.0 (0.0); m-xylene, 0.0 (0.08); o-xylene, 0.0 (0.0); p-xylene, 0.0 (0.0); toluene, 0.75 (1.78); arsine, 0.02 (0.13); PM_{10}, 37.9 (174); $PM_{2.5}$, 31.6 (154)	[1,5]‡
Fires involving residual recycling materials	Based on ambient monitoring carried out as part of the UK's AQinMI service	Incident-mean concentrations (ppb for gases and $\mu g\ m^{-3}$ for PM) with the maximum 1 min concentration in parentheses: CO, 0.37 (16.3); HCN, 0.13 (17.4); ammonia, 0.25 (10.5); HBr, 0.98 (31.3); HCl, 0.03 (7.02); HF, 0.05 (2.48); NO, 0.11 (0.48); NO_2, 0.44 (4.33); phosgene, 1.43 (9.23); SO_2, 0.06 (0.69); benzene, 0.2 (4.21); 1,3-butadiene, 0.27 (25); ethyl benzene, 0.11 (3.04); formaldehyde, 0.15 (4.96); methyl isocyanate, 0.2	[1,5]‡

		(1.45); m-xylene, 0.01 (0.86); p-xylene, 0.24 (4.91); toluene, 0.31 (9.02); arsine, 0.12 (0.9); PM_{10}, 136.9 (6141); $PM_{2.5}$, 79.1 (652)	
Landfill fires	Various controlled and uncontrolled landfill sites: the soils and vegetation adjacent to these sites were sampled for PAHs, and PCBs that had settled with the PM from the plume	Total PAHs ranged up to 300,000 $\mu g\ kg^{-1}$ dry weight (dw), though typically concentrations were in the thousands of $\mu g\ kg^{-1}$ (compared to reference concentrations ranging from 12 to 112 $\mu g\ kg^{-1}$. PCDD/Fs total PCB concentrations ranged from 0.2 to 7,900 $\mu g\ kg^{-1}$ dw.	[15]
Open-cut coal-mine fire in Latrobe, Victoria, Australia, February 2014	Burned for 45 days, creating a dense plume that affecting ca. 45,000 residents in local towns. The most affected location was the nearest town of Morwell, 0.5 km from the mine.	Concentrations given are ranges observed during fire at the most affected location: $PM_{2.5}$ 5.4-731 $\mu g\ m^{-3}$ for 24-h averaging period), CO (0-17.4 ppm); NO_2 (2-44 ppb), SO_2 (0-35 ppb); benzene (1.1-14 ppb for 24-h averaging period), toluene (0.5-4.8 ppm for 24-h averaging period), ethylbenzene (0.5-0.6 ppb for 24-h averaging period), xylenes (0.18-0.56 ppb for 7-d averaging period). 1,3-butadiene (0.5-2.5 ppb for 24-h averaging period), formaldehyde (1.4-7.6 ppb), B(a)P (0.1-8.2 $ng\ m^{-3}$).	[16]
Fire at International Terminals Company Deer Park Chemical plant, Harris County, Texas, USA	Fire affected several tanks containing naphtha and xylene and burned for 3 days.	benzene (highest recorded value of 32,000ppb in the industrial area, but measurements in the hundreds of ppb in other locations), isoprene (up to 1000 ppb), 1,3-butadiene (up to 1700 ppb), H_2S (69.7 to 119.2 ppb)	[4]
Fire at a naphtha cracking complex of a petrochemical complex in Yunling County, Taiwan in May 2011.	Fire burned for 10 hours. The site is adjacent to a residential area of 61,600 people.	Ehylene (57 ppb), propane (7 ppb), butane (6 ppb), toluene (2 ppb), benzene (21 ppb), vinyl chloride monomer (389 ppb), 1,3-butadiene (35 ppb).	[17]
Landfill fire in Sweden	Test site at which there was a controlled fire for monitoring purposes, but where an unplanned fire also broke out. Monitoring was carried out for both.	Total PAHs 810 $ng\ m^{-3}$, total PCBs, 590 $ng\ m^{-3}$, PCDD/Fs 2.88 to 20.54 $ng\ m^{-3}$ (0.051 to 0.427 $ng\ m^{-3}$TEQ).	[15,18,19]
Fire at a pool chemical manufacturing facility in Guelph, Ontario, Canada in August 2000	Initial explosion at the factory, followed by a fire that lasted for over 60-h	HCl, average over the survey period was 22 $\mu g\ m^{-3}$, with a maximum instantaneous value of 350 $\mu g\ m^{-3}$ and chlorine gas were monitored over the period of the fire. For Cl2 the average was 12 $\mu g\ m^{-3}$, with a maximum of 570 $\mu g\ m^{-3}$.	[20]
Fire at oil storage depot at Buncefield, Hertfordshire, UK in December 2005.	The fire burned for 5 days, though the high combustion temperature gave rise to a very buoyant plume that crossed to mainland Europe.	m-and p-Xylene 230 ppb, o-xylene 140 ppb, toluene 160 ppb, benzene 170 ppb, ethyl benzene 82 ppb, PM_{10} 1000 $\mu g\ m^{-3}$.	[1,5]
Fire at Bellolampo Landfill, Palermo, Sicily, Italy	Fire extended to a surface area of up to 120,000 m2 and lasted 18 days.	Ambient monitoring stations recorded an average of 50 $\mu g\ m^{-3}$ PM_{10} over the first 24-h, falling to 20 $\mu g\ m^{-3}$ thereafter. Elevated air and soil concentrations of dioxins were determined in soil exposed to the plume, as well as in milk	[21]

	samples in affected farms. Elevated heavy metals concentrations detected in soils.

‡including from supplementary material.

Finally, it is important to emphasise that major incident fires in urban environments do not only affect the quality of air in neighbouring communities but also cause contamination of soil and water environments. For example, e-waste fires have been reported to significantly elevate the soil concentrations of chlorinated and brominated PAHs [22]. Also, areas adjacent to landfill fires have been subject to significant PAH contamination, with total concentrations of up to 300,000 µg kg-1 dw in the worst case, but with typical concentrations in the thousands of µg kg-1 (compared to reference concentrations ranging from 12 to 112 µg kg-1 [15]. PCDD/F and polychlorinated biphenyl (PCB) contamination is also a significant issue with landfill fires, with concentrations up to 7,900 µg kg-1 dw being reported [15], together with elevated concentrations in milk, meat and high lipid-content vegetables such as olives [15]. Metal contamination can also be an issue with some fires, depending on the nature of the material that is burning [15]. Land contamination can additionally give rise to waterway pollution through leachate-runoff, and the waterways can be polluted directly by effluent from firefighting activities [23,24].

3. Monitoring and of Industrial Fires

There are numerous techniques and instruments that have been used for the determination of airborne contaminants for applications that include occupational health and hygiene, regulatory ambient monitoring, and stack emission monitoring. However, this review considers only those methods that have been deployed for monitoring during uncontrolled industrial fires. The reason is that there is likely to have been a selection process that has considered the implications for accuracy of the complex plume that is likely to be encountered in such circumstances [5].

As discussed in Section 2, there are likely to be hundreds of individual chemical substances present in the plume from a major industrial fire. These substances will be present in the gaseous phase and also in solid and liquid aerosols. The challenge of any public health response to a major fire is to have a monitoring equipment inventory that has sufficient techniques to facilitate the collection and measurement of those substances that represent the greatest threat to the short- and long-term health of affected populations. The choice of monitoring technique is a compromise between providing an appropriate temporal resolution for often rapidly developing emergency situations (dependent on meteorology, e.g., wind direction and the development of the fire itself) and the need to accurately characterise the components and their concentrations within the plume [1]. Often, portable sensors are used because they will be able to provide near real-time information on airborne pollutant concentrations to the relevant public health responder, facilitating timely decision making on the appropriate public health intervention, e.g., to shelter indoors or to evacuate. The downside of portable monitors is that they tend to have higher detection limits (typically ppm for VOC/VIC analysis), lower accuracy and lower specificity compared to reference methods. Nevertheless, as long as these limitations are well characterised and understood, decisions on mitigation can be made with reasonable confidence.

Table 3 gives an overview of the range of monitoring techniques that have been used to measure pollutant concentrations during major industrial fires, along with detection limits where known. In the UK, the AQinMI service employed both portable continuous monitors (Osiris laser light scattering for PM, Gasmet FTIR (Fourier Transform Infrared) for VOCs and VICs, electrochemical for chlorine gas and carbon monoxide) and collection devices for later laboratory analysis (impinger solutions, filters, silica gel, thermal desorption tubes and PUF filters) [5]. Evaluations carried out on monitoring results obtained from actual industrial fires showed that PM measurements using the Osiris could be considered reliable because of the good agreement between monitored results obtained during co-location studies, as well as having single µg m^{-3} monitoring resolution [5]. For VICs and VOCs using the Gasmet continuous monitor, relative emission profiles matched those expected for the particular type of fire being monitored, e.g., tyre or other types of waste fire [1]. Also, reported concentrations for most substances were consistent with the results from other similar monitoring studies [5]. However, acrolein and phosphine were found to be reported at unrealistic concentrations, i.e., above limits that would result in serious health effects, when none were actually reported [1]. Cross-interference between the FTIR spectra of acrolein and phosphine and other, uncharacterised, substances in the plume was the likely explanation for the over-reporting (the FTIR has 24 pre-set substances for which there is calibration and correction for interference effects, but if an unknown substance is present in high concentrations, any overlap in spectra will not be accounted for). Nevertheless, the portable continuous monitoring techniques discussed above were used in practice as the primary source of data on which to base public health response decisions for more than 30 major industrial fires that had the potential to affect significant numbers of people in nearby communities [1,5].

Electrochemical cell-based devices have been deployed for some industrial fires. Typically, there will be an individual cell for each component being monitored, and whilst they have poorer detection limits than reference methods and are prone to cross interference for example NO_2 and ozone [25], they have been found to be useful for the concentrations encountered during fires, particularly when monitoring firefighter exposure [26]. Photoionisation devices offer good detection limits for total VOCs but cannot differentiate between different VOC species and have a further disadvantage that some VOCs cannot be ionised at the energy emitted by the lamp leaving them unaccounted for [27]. Nevertheless, these devices can provide valuable information for an initial survey of concentrations during an incident [28]. Additionally, photoionisation data combined with information on the source material of a fire, and likely main pollutants [1,13,14] could provide a sufficient basis for an initial health risk assessment to be made.

The most accurate routine technique for determining VOCs is GC-MS, and in Table 3 there are examples of the use of both continuous automatic GC systems, as well as sampling followed by laboratory analysis. Continuous systems are generally associated with fixed location monitoring stations because of the size of the instrument and the requirement for services, as was the case with the Deer Park fire in Houston [28], although these types of instrument can be placed in monitoring vans [20]. A more convenient approach is to collect the samples in vacuum cannisters [26,34] or on adsorbent tubes (pumped or passive) and returned to the lab for analysis [31], although this delays the forwarding of monitoring information to the public health responders.

Table 3. Overview of monitoring techniques reported in the literature to measure pollutants emitted from industrial fires.

Monitoring technique and principle	**Context for monitoring**	**Determinands, with detection limits (dl) in parentheses, where available (in ppm, or ug/m3 for particulates).**	**Reference**
Continuous / real time techniques			
Laser light scattering (670nm) (Turnkey Osiris particulate monitor)	Included in equipment inventory for deployment to UK AQinMI fires.	Total suspended solids (TSP), PM_{10}, $PM_{2.5}$, and PM_1 (all 0.1).	[5,29]
Low cost particulate sensors	Wildfires in california	$PM_{2.5}$ PM_{10}. See reference for performance characteristics.	[30]
Beta Attenuation monitor	Fire at open-cut coal min in Latrobe, Victoria, Australia.	$PM_{2.5}$ (3.4 for 1-h)	[31,32]
Infrared (Gasmet DX4030/40)	Included in equipment inventory for deployment to UK AQinMI fires	Carbon dioxide (10), carbon monoxide (1), nitrous oxide (0.02), methane (0.11), sulfur dioxide (0.30), ammonia (0.13), hydrogen chloride (0.20), hydrogen bromide (3.0), hydrogen fluoride (0.2), hydrogen cyanide (0.35), formaldehyde (0.09), 1,3-butadiene (0.20), benzene (0.3), toluene (0.13), ethyl benzene (0.08), m-xylene (0.12), o-xylene (0.12), p-xylene (0.12), acrolein (0.25), phosgene (0.2),	[1,5]

		arsine (0.02), phosphine (0.2) and methyl isocyanate (0.25).	
Proton Transfer Reaction Time-of-Flight Mass Spectrometry (PTR-TOF-MS)	Monitoring of a biomass fire	132 separate VOCs (single ppb)	[11]
Automated gas chromatograph	Fire at International Terminals Company, chemical factory, Deer Park, Houston, Texas.	Range of VOCs and other hazardous air pollutants (HAPS) (0.4 ppb-C)	[4]
Electrochemical cell (GFG-Microtector II G460)	Industrial fires in Saudi Arabia (exposure by firefighters)	Carbon monoxide (1), hydrogen cyanide (0.5), ammonia (1), sulfur dioxide (0.1), hydrogen chloride (0.2), Hydrogen sulfide (0.2 ppm).	[26]
QRAE electrochemical cell	Included in equipment inventory for deployment to UK AQinMI fires	Chlorine (dl) and carbon monoxide (dl)	[5]
ppbRAE photoionization detector	Fire at International Terminals Company, chemical factory, Deer Park, Houston, Texas.	Total VOCs (ppb)	[28]
Jerome gold film electrical resistance analyser	Included in equipment inventory for deployment to UK AQinMI fires	Hydrogen sulfide (3 ppb)	[5,33]
Triple quadrupole mass spectrometer Trace atmospheric gas analyzer (TAGA IIe)	Chemical works (chlorine-based pool chemicals), Guelph, Ontario, Canada	HCl and Cl_2 (0.5 µg m^{-3})	[20]
Mobile photochemical Monitoring station (MPAMS) and open-path FTIR	Fire at a naphtha cracking complex of a petrochemical complex in Yunling County, Taiwan in May 2011.	Ehylene, propane, butane, toluene, benzene, vinyl chloride monomer, 1,3-butadiene, all at ppb levels.	[17]
Non-continuous techniques			
Tecora Delta Low flow pump (1 L min^{-1}) with impinger: absorption into solution followed by wet chemical analysis	Included in equipment inventory for deployment to UK AQinMI fires	Hydrogen cyanide, acetic acid, hydrogen sulfide, chromic acid (impinger solution of 0.05M sodium hydroxide). Ammonia (impinger solution of 0.05M sulfuric acid)	[5]
Tecora Delta low flow pump pump (1 L min^{-1}) with PTFE filter + silver membrane	Included in equipment inventory for deployment to UK AQinMI fires	Bromine and chlorine (dl dependent on sampling period).	[5]
Tecora Delta low flow pump (0.5 L min^{-1}) with Silica gel – Supelco Orbo 53	Included in equipment inventory for deployment to UK AQinMI fires.	Hydrogen fluoride, nitric acid, phosphoric acid, sulphuric acid, sulfur trioxide and arsine (dl dependent on sampling period).	[5]
Tecora Delta low flow pump (0.2 L min^{-1}) with thermal desorption (TD)	Included in equipment inventory for deployment to UK AQinMI fires	(All at ppb levels) 1,1,1-trichlorothane, 1,2-dichloroethane, 1,3-butadiene, 2,4-toluene diisocyanate, 2,6-toluene diisocyanate, acetone, acetonitrile, acrolein, acrylamide, acrylonitrile, benzene, CS2, chlorobenzene, chloroform, chloropicrin, dichloromethane, ethyl acrylate, ethyl benzene, ethyl isocyanate, ethylene oxide, formaldehyde, methyl acrylate, methyl bromide, methyl chloride, 2-butanone, methyl isocyanate, methyl isothiocyanate, methyl methacrylate, methyl styrene, phenol, phosgene, propane, styrene, tetrachloroethylene, tetrachloromethane, toluene, trichloroethylene, vinyl chloride, xylene, other volatile organic compounds (dl dependent on sampling period).	[5]

Tecora Delta low flow pump (10 L min^{-1}) with gridded asbestos filter	Included in equipment inventory for deployment to UK AQinMI fires	Asbestos	
SUMMA 6-L vacuum canister for collection, followed by GC-MS	Industrial fires in Saudi Arabia (exposure by firefighters)	(All at ppb levels) 1,3-butadiene, acetone, trichloromonofluoromethane, 1,1-dichloro-ethene, methylene chloride, carbon disulfide, methyl tert-butyl ether, 1,2-dichloro-ethene, benzene, bromodichloromethane, methyl isobutyl ketone, heptane, toluene, tetrachloroethylene, ethylbenzene, m-xylene, o-xylene, p-xylene, styrene, 1,3,5-Trimethylbenzene, benzyl chloride and 1,2,4-Trichlorobenzene.	[26]
SUMMA 6-L vacuum canister for collection, followed by GC-MS	Buncefield oil storage fire)	(All at ppb levels) m- and o- and p-xylenes, toluene, benzene and ethyl benzene.	[34]
Cannister followed by GC-MS	Fire at open-cut coal mine in Latrobe, Victoria, Australia	Speciated VOCs (All at ppb levels)	[31]
Radiello diffusive sampler with adsorbent (modified scintered microporous polyethylene)	Fire at open-cut coal mine in Latrobe, Victoria, Australia	Speciated VOCs (All at ppb levels)	[31]
2,4-dinitrophenylhydrazine (DNPH)-coated solid sorbent cartridges, collecting carbonyls as derivatives, followed by elution and analysis by high-performance liquid chromatography (HPLC).	Fire at open-cut coal mine in Latrobe, Victoria, Australia	Carbonyl compounds (e.g., formaldehyde, acetaldehyde, acrolein, acetone and benzaldehyde.	[31]
Tecora Echo high volume sampler (200 L min^{-1}) with quartz filter. Gravimetric with extraction from filter.	Included in equipment inventory for deployment to UK AQinMI fires	Antimony, arsenic, cadmium, chromium, lead, manganese, nickel, platinum, thallium, vanadium, mercury, other metals, polycyclic aromatic hydrocarbons, polychlorinated biphenyls, pesticides	[5]
Polyurethane Foam (PUF) filters	Fire at open-cut coal mine in Latrobe, Victoria, Australia. Also included in equipment inventory for deployment to UK AQinMI fires	Dioxins and derivatives, including polychlorinated dibenzodioxins, furan and derivatives, including polychlorinated dibenzofurans. PAHs	[5,31]

The selection of sampling location is also of critical importance since the monitoring data should ideally reflect public exposure. For some incidents, such as the Deer Park chemical works fire [4,28], existing monitoring networks were utilised to provide contemporaneous monitoring data. Moreover, with the increasing use of low cost sensors [35], it is likely that many future incidents may have readily available data from arrays of such sensors located in neighbouring areas. Nevertheless, there remains a requirement for a mobile monitoring capability, such as the UK's AQinMI scheme [5].

4. Modelling of Public Exposure to Harmful Airborne Pollutants from Industrial Fires

Monitoring data, whilst providing real-world concentrations of harmful airborne pollutants released during industrial fires, is limited in terms of the spatial resolution of the monitoring and the choice of measurement locations. It is difficult to predict the exposure profile of the population from monitoring data alone, and so there is a requirement for plume modelling to support the estimation of likely exposure. There are a range of different models and approaches used, with some notable literature examples of the modelling of dispersion of pollutants from fires and chemical accidents detailed in Table 4. For example, in the UK, the Chemical Meteorology (CHEMET) service provides a semi-quantitative estimate of the plume path and to identify populations that are likely to be exposed. Timely precautionary public health advice can be provided on the basis of the CHEMET plume [5].

Table 4. Review of studies that have modelled the release and dispersion of air pollutants from industrial fires or incidents.

Dispersion models (categorised by type)	Summary of study and results	Ref
Gaussian		
ADMS (Atmospheric Dispersion Modelling System)	This was the simulation of the release of pollutants from a chemical warehouse fire that was contained within the building and where the emissions from a buoyant plume were made through roof vents and doors. The effect of the ADMS building module was studied. The results were compared to wind tunnel tests.	[36]
ADMS	Simulation of a fire at a car recycling plant. Particular focus was on parameters used, e.g., source temperature, area, exit velocity and emission rate. The best performance against monitoring data was for an area source.	[37]
UK CHEMET	Routinely used, particularly in the early phase of a fire to track the dispersion of plumes from fires or chemical releases. Basic model that calculates plume path. Larger incidents may also be modelled using NAME (see below).	[1,5]
EPA Screen Model; 3	Simulation of worst case scenario for ground level concentrations of pollutants from an uncontrolled tyre dump fire (mg m^{-3}): CO (674), PM (652) and PAH (24) concentrations 500m downwind of the fire.	[38]
General Gaussian plume model after first calculating the plume rise.	The study aimed to simulate concentrations of airborne contaminants in adjacent areas in the case of a fire at a chemical and pharmaceutical plant in Denmark. A range of different meteorological conditions were modelled.	[39]
CFD		
CFD code coupled with a mesoscale metreological model	Simulation of PM emission during a hypothetical industrial fire. The CFD component was used to accurately parameterize the fire and developing plume, which was then modelled on a meso-scale using the meteorological model. The results were compared to a system where the CFD component was replaced by the ForeFire model.	[40]
CFD-RANS (Computational Fluid Dynamics Reynolds-Averaged Navier Stokes)	CFD-RANS was used to calculate dispersion for the accidental release of 900Kg of vinyl chloride monomer at an industrial facility. Modelling was restricted to the chemical facility boundaries, comparing the predicted concentrations with real monitoring data. Performance was considered satisfactory, especially given that there were uncertainties in wind direction.	[41]
Eulerian		
TAPM (The Air Pollution Model, CSIRO), combined with a Chemical Transport Model	Open-cut coal mine fire in Latrobe, Victoria, Australia. Allowed the calculation of personalized mean 24-h (0-56 μg m^{-3}) and peak 12-h (0-879 μg m^{-3}) PM$_{2.5}$ exposure for a range of participants who were also asked about health effects of the fire.	[6]
Lagrangian		
HYSPLIT (Hybrid Single Particle Lagrangian Integrated Trajectory)	Fire at a naphtha cracking complex of a petrochemical complex in Yunling County, Taiwan in May 2011. Used back-trajectory model to trace monitored pollutants (propane, butane, toluene, benzene, vinyl chloride, 1,3-butadiene to the source of the fire from distances up to 10km.	[17]
HYSPLIT	Fire following bombing of industrial facilities in Serbia during the Kosovo war. Used in back-trajectory mode to identify the likely source of elevated concentrations of aerosol-bound PAHs, dioxins, and furans that were detected in Greece.	[42]
HYSPLIT	Modelling dispersion of pollutants at a fire involving six tanks of a 179-tank fuel storage facility in port of Santos, Brazil. Used in forward mode to calculate trajectories for the fire, with results showing that the plume was mainly carried out to sea, explaining the relatively small increases in PM$_{10}$ observed at monitoring stations in the city.	[24]
HYSPLIT and PyTREX	Accidental release of methyl mercaptan from a chemical works in Rouen, France. Unknown emission rate, so a nominal concentration was modelled in forward mode to try and explain the very geographically and temporally separated reports of odour nuisance (in Paris and London and over a period of more than 24 hours). Both models gave similar results.	[43]
HYSPLIT	Fire at a chemical plant in Houston, Texas. Used in forward mode to predict a range of possible trajectories of the plume and identify which monitoring stations would have been subject to elevated airborne pollutants from the fire.	[4]

CALPUFF and SPRAY	Hypothetical refinery fire. Sensitivity analysis of the key model parameters that might affect the calculation of ground level concentrations of a (tracer) gas. Of the model parameters tested (source diameter, temperature, height and exit velocity), diameter was found to be the most sensitive.	[44]
CALPUFF	Burning of wood waste from trees killed by the Mountain Pine Beetle, British Columbia, Canada. Modelled the meteorological conditions, and distances from the centre of the city of Prince George, where wood burning could be permitted (based upon exceedances of the Canada standard for $PM_{2.5}$).	[45]
ADMS-Star	Tyre fire at Mexborough, UK in 2010. Used back-calculation method to estimate emission rates. Predicted 24-h average concentrations for those exposed to the plume. Estimated that 7856 residents may have been exposed to PM_{10} concentrations in the US EPA AQI category of 'Hazardous' (>425 μg m^{-3}).	[7]
NAME	Fire at fuel storage site (20 tanks) at Buncefield, Hertfordshire, UK. Used a nominal concentration of tracer gas in the model to allow calculation of the dispersion in three dimensions. Accurately modelled plume rise and geographical spread. Corroborated by satellite imagery.	[46]

For more realistic modelling of ground level air pollutant concentrations, there are a range of options, including Gaussian, Lagrangian (puff and particle), Eulerian and Computational Fluid Dynamics (CFD) methods [43,44,47,48]. Modified Gaussian models, such as ADMS and ADMS-Urban are ideal for regulatory compliance work, but the assumption of a steady state across the modelling domain may be problematic for meso-scale dispersion calculations required for plume transport simulation over large areas [44]. Nevertheless, there are literature examples of the ADMS being used to simulate downwind emissions from fires [36,37], and it is stated that ADMS is suitable for modelling areas in the tens of km [37]. Other Gaussian approaches to the modelling of fires have also been reported [39]. Eulerian models, often operating on regional or continental scales, use a fixed grid, solving partial differential equations for transport and chemical reactions for each grid cell. Eulerian approaches can be considered as advanced simulators of dispersion and have been applied to an open-cut coal mine fire in Australia [6], though applications may be limited because there is a requirement for significant computational resources [44]. CFD approaches are also very computationally expensive [40], but provide the highest resolution modelling of all systems and are ideal for simulating dispersion over small areas (< 1km) that may have complex terrain and infrastructure. An example is the CFD modelling of the accidental release of vinyl chloride on a small geographical scale (within the boundary of a chemical works) [41]. One method of utilising CFD in wider scale dispersion calculations is in a coupled model, whereby CFD is used to model the fire conditions and initial development of the plume, followed by mesoscale meteorological modelling of dispersion over a wider area [40].

By far the most widely used approaches to the modelling of plume dispersion from fires are the Lagrangian methods, including, CALPUFF (California PUFF model), HYSPLIT (Hybrid Single Particle Lagrangian Integrated Trajectory), NAME (Numerical Atmospheric-dispersion Modelling Environment, as used by CHEMET) and ADMS-Star. Langrangian methods calculate the trajectories of discrete 'puffs' or groups of 'particles' of pollutants as they are influenced by meteorological conditions. These models can be used in backwards-mode to help in the identification of sources of elevated pollution concentrations [17,42] or odour [49]. They can also be used in forward mode to either calculate ground level concentrations and overall plume behaviour [46] or identify the ambient monitoring stations that a plume is likely to have passed over, thus allowing incident-relevant data on airborne pollutants to be identified [4,24,43].

One problem associated with all these approaches is the calculation of an accurate source emission rate. Often, because of the considerable uncertainty in this parameter, nominal emission rates are modelled, allowing a semi-quantitative evaluation of those areas that are likely to be affected and where the highest concentrations are expected. An example is the modelling of the complex and geographically disperse pattern of odour complaints subsequent to a methyl mercaptan release in France [43].

There are methods for calculating plume components and their emission rates, based on the composition and volume/area of the fire source material, together with data from standard combustion studies, and this has been used as a basis for modelling waste fires [37,45]. An alternative approach is to back-calculate emission rates, based on triangulated observations of monitoring data at several locations. ADMS-Star, a Lagrangian puff model, has the facility to make such calculations and this has been applied to a large tyre fire in Mexborough, UK, which was attended by two monitoring teams from the UK AQinMI scheme. ADMS-Star calculates an emission rate that fits the observed pollutant concentrations and meteorological conditions, and subsequently forward-models the evolution of the plume. Predicted ground level concentrations can be combined with population data to estimate an overall exposure profile [7]. Other approaches to estimating the exposure profile of affected populations have been reported, for example during a long-established fire in an open-cut mine [6].

5. Health Impacts of Major Incident Fires

As we have seen, reported airborne pollutant concentrations during major incident fires can reach levels that can be an order of magnitude higher than ambient standards, particularly for PM, which is the main focus of this section.

Short-term (24-h or less) air quality guideline values and AQI category boundaries for PM are based on epidemiological data on the effects of air pollutants on human health [50] and so exceedance of these levels indicate that adverse health effects are likely to occur, particularly for the most vulnerable members of the population (in terms of age and pre-existing health conditions). The magnitude of expected health effects within an exposed population can be estimated from established epidemiologically-derived risk factors for exposure. Thus, for PM_{10} there is a 0.41% increase in short-term mortality for every 10 µg m^{-3} increase in concentration (for $PM_{2.5}$ the risk factor is and 0.65%)[51] and for all-cause hospital admissions there is a 0.18% increase for every 10 µg m^{-3} increase in PM_{10} concentration [52].

Whilst short-term effects of PM have been well characterised for periods of 24-h and above, it is known that physiological responses to PM can be observed over a much shorter exposure timeframe, i.e., a few hours. For example, upper respiratory tract inflammation has been observed in human volunteers exposed to diesel fumes containing PM_{10} concentrations ranging from 100 to 300 µg m^{-3} for exposure periods as short as 1 to 2 hours [53]. In addition, firefighters who are frequently exposed to high PM concentrations show evidence of inflammatory response after shifts [54,55], with pro-inflammatory proteins IL6 and IL8 remaining elevated in the bloodstream for at least 3 months after exposure [54].

Given the epidemiological and physiological evidence, we must conclude that the concentrations of PM reported in some incidents detailed in Table 2 would be expected to have had a short-term health impact on the exposed population, though evidence for the capture of these effects by syndromic surveillance (e.g., hospital emergency room admissions) is not clear cut. For example during the Buncefield oil storage fire in 2005, and the Chancery Lane fire in London in 2009, monitoring of doctors' surgery visit data and calls to the UK National Health Service-Direct indicated that there were no detectable population-based health effects such as coughs, breathing difficulties, asthma admissions or respiratory infections in those communities most at risk from exposure [56]. Similarly, a syndromic surveillance programme carried out in the aftermath of arson-initiated landfill fires in Palermo, Sicily, did not register any increase in daily hospital admissions for respiratory diseases or other related health effects [21]. However, there is evidence of respiratory health effects from self-reported outcomes after an open-cut coal mine fire in Australia [6]. The study, which used individualised $PM_{2.5}$ exposure data for participants, derived from modelling studies, found increases of 13% and 10% respectively for chronic cough and chronic phlegm per 10 µg m^{-3} increase in $PM_{2.5}$ [6].

For PM exposure in the related field of wildfire research, health effects are well characterised [57–60] including for the more vulnerable members of the exposed population [61,62]. It is also important to emphasise that combustion-derived particulates have been found to be more toxic than ambient particulates due to the presence of a wide range of chemical toxins, including heavy metals and PAHs [63].

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For pollutants other than PM, Tables 1 and 2 show monitoring data for PAHs, PCDD/Fs, PCBs, metals, VICs and VOCs. Exceedances have been noted for several of these substances, for example benzene during the Deer Park fire in Texas [4]. In a study of ground-level concentrations of organic and inorganic substances during some 34 major incident fires, Griffiths, et al. [1] found exceedances of ambient short-term guidelines for NO_2 (14 incidents exceeded the 0.11 ppm 1-h EU guideline value) and SO_2 (0.13ppm EU 1-h guideline value was exceeded in 15 incidents; WHO 10 minute guideline of 0.19 ppm exceeded in 21 incidents; and UK 15 minute guideline of 0.10 ppm exceeded in 30 incidents). For other substances, HCN, HBr, HF and formaldehyde exceeded the Level 1 US Acute Exposure Guideline Levels (AEGLs), for which temporary non-disabling health effects might be expected [1].

Nevertheless, as previously discussed, incident monitoring is often carried out though networks of ambient air quality measurement stations. Due to prevailing meteorological conditions during such incidents, these stations may not necessarily be in the best location to identify where communities are exposed to the highest ground level concentrations. The advantage of a coordinated mobile monitoring network, such as the UK's AQinMI scheme is that monitoring stations can be set up in locations where concentrations are predicted to be high, therefore allowing a more realistic estimation of population exposure, especially if combined with modelling.

One final point is that health effects may be psychological in nature, with the potential for stress and anxiety from feared exposure to the plume, and also, if necessary, from the instruction to shelter [9,64,65].

6. Suitability of Guideline Values for PM

As discussed in previous sections, PM is one of the most health-impacting air pollutants to which communities adjacent to industrial fires are exposed. During such incidents, hourly and daily concentrations of PM_{10} and $PM_{2.5}$ can be in the high hundreds of $\mu g\ m^{-3}$, with some incidents even recording concentrations in the thousands of $\mu g\ m^{-3}$ for PM_{10} [50]. And yet we also know that physiologically measurable effects of exposure, such as inflammatory response, can be evident after exposure periods of just hours [53–55,66].

Historically, short-term guideline values (GVs) for PM, such as those developed by the WHO, the US EPA, and the European Union, have been derived from epidemiological evidence of health effects over a 24-h period [66]. And whilst it has been acknowledged that health effects of PM exposure can manifest in much shorter timeframes, the notion of setting limits at averaging times as low as 1-h, as with ozone, has been dismissed because of the significant correlation between the concentrations of the 1-h maximum (in a 24-h period) with the 24-h average [66]. This said, there is some literature evidence that daily mortality [67] and COPD hospital admissions for over 65s [68] are more strongly associated with the daily maximum 1-h concentration rather than the 24-h average, suggesting that a 1-h GV might be beneficial [69,70].

However, the observation of a correlation between the concentrations of the 1-h maximum (in a 24-h period) with the 24-h average, does create an opportunity to predict exceedances of 24-h guidance based on 1-h concentrations. Such predictions would allow public health decisions, for example on whether to advise affected communities to stay indoors or evacuate, to be made at a much earlier stage, and so avoid the need to wait 24 hours to confirm a breach, which could be potentially harmful, particularly for those who are more vulnerable. Offering a 1-h short-duration PM exposure guideline aligns with the offering of organic and inorganic species guidelines as outlined in AEGLs [71] and ERPGs [72].

Several approaches have been used in the literature to derive *de facto* 1-h guideline values for both PM_{10} and $PM_{2.5}$, as listed in Table 5. For Europe's Common Air Quality Index (CAQI), 1-h thresholds for AQI category boundaries for PM_{10} were derived by dividing the 24-h concentration by a factor of 0.55; this is the ratio of the 24-h average concentration and the maximum 1-h concentration in the same period, based on ambient monitoring data from 52 sites throughout Europe over the period 2001 to 2004. 1-h values for $PM_{2.5}$ were obtained by multiplying the 1-h PM_{10} AQI boundary thresholds by a factor of 0.6, which was the observed fraction of PM_{10} that is $PM_{2.5}$. Stieb, et al. [73], using Canadian monitoring data, obtained a similar PM_{10} ratio to that used by CAQI, though they used a maximum 3-h concentration in a 24-h period.

Table 5. Summary of derived 1-h GVs for PM.

Limit or guideline	24-h concentration / μg m^{-3}	Derived 1-h concentration / μg m^{-3} (if available)
PM_{10}		
CAQI[‡] Low	12	25[‡]
CAQI Medium	25	50[‡]
CAQI High	50	90[‡]
CAQI Very High	100	180[‡]
WHO AQG[*]	45	-
WHO Interim Target 4 / EU 24-h guideline	50	-
WHO Interim Target 3	75	109[†]
WHO Interim Target 2	100	146[†]
WHO Interim Target 1	150	
US EPA AQI 100: Unhealthy for sensitive groups	155	227[†]
US EPA AQI[Ψ] 150: Unhealthy	255	427[†]
US EPA AQI 200: Very unhealthy	355	707[†]
US EPA AQI 300: Hazardous	425	945[†]
UK threshold for evacuation	320	510[†]
$PM_{2.5}$		
CAQI Low	10	15[‡]
CAQI Medium	20	30[‡]
CAQI High	30	55[‡]
CAQI Very High	60	110[‡]
WHO AQG	15	-
WHO Interim Target 4 / EU 24-h guideline	20	-
WHO Interim Target 3	30	48[†]
WHO Interim Target 2	50	63[†]
WHO Interim Target 1	70	93[†]
US EPA AQI 100: Unhealthy for sensitive groups	35	45[†]
US EPA AQI 150: Unhealthy	55.5	70[†]
US EPA AQI 200: Very unhealthy	150.5	183[†]
US EPA AQI 300: Hazardous	250	345[§]

[‡] Van den Elshout, et al. [74], Airly [75]. [†] Deary and Griffiths [50]. For 99% True Positive Rate. [§] Deary and Griffiths [50]. For 100% True Positive Rate. [*] WHO [51]. [Ψ] US Federal Register [76], Perlmutt and Cromar [77].

However, being based on ambient monitoring data, the CAQI range ('Very Low' to 'Very High') of 0 - 100 μg m^{-3} and 0 – 180 μg m^{-3} for 24-h and 1-h boundary thresholds respectively for PM_{10} (0 – 60 μg m^{-3} and 15 – 110 μg m^{-3} respectively for $PM_{2.5}$), is low compared to the typical concentrations recorded in major incident fires. Also, the descriptors employed by CAQI are not directly related to health. A more appropriate AQI for the magnitude of concentrations observed in industrial fires, wildfires and dust storm is the US EPA AQI, which ranges to 500 μg m^{-3} for $PM_{2.5}$ and 605 μg m^{-3} for PM_{10}, and which employs health based descriptors such as 'unhealthy' and 'hazardous' (see Table 5). Deary and Griffiths [50] have employed Receiver Operating Characteristic (ROC) statistical analysis, based on 38 million rolling 24-h periods of monitoring data from across the US to derive 1-h threshold concentrations that predict exceedances on the US EPA AQI classification boundary concentrations. The resultant 1-h threshold concentrations listed in Table 5 for both $PM_{2.5}$ and PM_{10} will correctly predict 99% of all exceedances of the corresponding 24-h guidelines (99% True Positive Rate, TPR), based on the overall US dataset, though other TPRs, e.g., 100% or 95% can be chosen, thus allowing some flexibility with respect to balancing public perception of the number of false alarms with accuracy of predicting exceedances. In addition, the same authors have derived 1-h GVs for some of the WHO [51] Interim Targets and for a UK 24-h trigger to evacuate [78], which was similar to the value obtained from an earlier analysis on the same threshold, based on AQinMI data [5].

1-h GVs, as listed in Table 5, allow public health responders to protect exposed populations against the immediate effects of episodic air pollution from industrial fires, allowing timely guidance to be communicated, and appropriate mitigating measures to be actioned.

7. Environmental Justice and Socio-Economic Factors

Industrial sites are often located in close proximity to lower-income neighbourhoods, which may also have a higher representation of minority ethnic populations as well as poorer overall levels of health and lower overall educational attainment than the general population [4,79–81]. Routine emissions from such sites and the potential for accidental releases and fires, mean that economically disadvantaged and minority populations may be disproportionately exposed to airborne environmental hazards [4]. Moreover, such communities, might not have access to the resources and expertise needed to advocate for better environmental standards nor for taking mitigating measures such as relocation or personal protection [7]. Several of the fires listed in Table 2 concern issues of environmental justice, including the chemical fire at Deer Park [4] and landfill fires in Palermo [21]. In the UK, the 2010 waste trye fire at Mexborough was in very close proximity to neighbourhoods that had higher levels of deprivation and a greater proportion of individuals with pre-existing health conditions [7]. A range of actions have been suggested to reduce inequities [4,15,21]. Thus, for communities adjacent to existing sites, there is a requirement for: continuous airborne pollutant monitoring, both at the industrial sites and within the communities; improved public health communication during and after incidents; clear emergency protocols; syndromic surveillance monitoring protocols for affected populations; resources for post-incident monitoring of land, water, foodstuffs and livestock; enhanced regulation; and active community engagement in incident investigation and in raising awareness of potential health risks within the communities [4,15,21]. For future industrial developments, there is a requirement for clear zoning so that emissions and potential accidental releases will be well away from communities [4,21].

8. Conclusions, Recommendations and Research Needs

Airborne pollution from major industrial fires in urban settings is a significant yet under-researched public health hazard that disproportionately affects those from minority ethnic groups, those with underlying health conditions and those from disadvantaged and socio-economic groups. Ground level concentrations in such fires often exceed short-term health-based guideline values, and there is a risk of contamination of soil, water, crops and livestock from dry and wet deposition of pollutants and from firefighting activities.

Our review has highlighted the importance of having a robust monitoring capability that can be deployed during such incidents, and which can accurately characterize the most health-impacting pollutants and their concentrations. Moreover, there is a requirement for the provision of near real-time monitoring data to public health responders so that timely advice can be communicated to affected populations. However, due to the complexity of the plume and the number of individual chemical species present, there may be issues of cross-interference for some continuous techniques, for example, electrochemical or FTIR methods. Research is therefore required to enhance monitoring capabilities in preparation for major incident fires, evaluating techniques such as direct air sampling mass spectrometry (DS-MS) and differential optical absorption spectroscopy (DOAS) [82], and how such systems, which are often bulky and require significant services, might be made portable and deployed in practice [82,83]. The utilization of such equipment would require mobile monitoring vans to be available at key locations throughout a country or territory but would also require significant maintenance and calibration. Alternatively for more portable methods, such as FTIR gas analysis that are often currently employed [1], advanced machine learning or AI approaches could be researched for accurate characterization of individual plume components within the overall mix [84]. PM is probably the most significant health hazard from industrial fires and it is reassuring that portable monitoring systems, including low-cost sensors, have an acceptable level of accuracy [85,86] and can be deployed with good spatial resolution, as is the case in the US for wildfires [87]. Networks of sensors could be employed surrounding those industrial sites that are most prone to fires, as well as in adjacent communities, possibly also combining monitoring information with remote sensing to estimate the plume extent and overall population exposure [88]. The use of drones fitted with low-cost sensors, perhaps as swarms, could offer an alternative to characterising the plume extent, including in the vertical axis, helping estimate exposure for those living in high-rise buildings [89,90]. Modelling can also be utilised to accurately characterise plume concentrations, though the calculation of the emission rate of the fire remains a source of uncertainty [7]. This could possibly be overcome with higher spatial resolution monitoring data combined with back-calculation methods [7]. Additionally, research into coupled CFD/meteorological modelling and other advanced approaches may produce more accurate plume predictions [40].

The health impact of major incident fires is another key area for further research, especially given the high reported concentrations of airborne pollutants in many incidents, particularly for PM. Research is needed to identify the most appropriate syndromic surveillance approaches for industrial fires, both for future events but also in retrospectively evaluating health impacts. There is also a case for carrying out retrospective epidemiological studies on longer-term effects of industrial fires, where appropriate health data can be identified. Health effects should be investigated within a context of environmental justice, thus accounting for ethnicity, educational attainment, employment, socio-economic status and underlying health status.

Research is also needed into preventative measures, mitigation and the public health response. Preventative measures should focus on the planning system so that appropriate zoning of industrial and residential areas is in place. In addition, where industrial facilities are already adjacent to residential areas, regulatory and technical controls should be reviewed, together with the installation of appropriate monitoring technology. Community involvement should be encouraged where possible [4]. For incident response, there are exemplars from different countries, such as the UKs AQinMI service and the associated public health response [1,5,9,91,92]. Finally, to aid in the public health response to industrial fires, and the timeliness of decision making, 1-h guideline values for PM could be considered [50].

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