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Review

Wildfire smoke and public health risk

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Abstract. Wildfire activity is predicted to increase with global climate change, resulting in longer fire seasons and larger areas burned. The emissions from fires are highly variable owing to differences in fuel, burning conditions and other external environmental factors. The smoke that is generated can impact populations spread over vast geographical areas. Wildfire smoke is a complex mixture of pollutants that can undergo physical and chemical transformation processes during transport and can have major impacts on air quality and public health. This review looks at the main features of smoke that should be considered in the assessment of public health risk. It describes the current state of knowledge and discusses how smoke is produced, what factors affect emissions and smoke distribution, and what constituents of smoke are most likely to cause adverse health effects.

Additional keywords: climate effects on emissions, forest fires, health impacts, mercury, particulate matter, smoke production.

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Introduction

Concern about the potential impacts of smoke events on public health is growing. Public health risks have been documented during recent large wildfire events (Delfino *et al.* 2009; Hänninen *et al.* 2009; Wegesser *et al.* 2009; Henderson *et al.* 2011; Rappold *et al.* 2012; Betha *et al.* 2014) and there is mounting evidence that fire activity may increase in some parts of the world, such as the western USA, Canada, Australia, Russia and Europe (Flannigan *et al.* 2009; Spracklen *et al.* 2009; Keywood *et al.* 2013). These predicted increases are attributed to extreme weather conditions, forest management practices, climate change and a rise in human population density near fire-prone areas.

Wildfires can release large amounts of particulate matter (PM) and toxic gases including carbon monoxide (CO), nitrogen oxides (NO_x) and non-methane organic compounds (NMOC) into the atmosphere. Emissions from wildfires are composed of primary pollutants (e.g. PM, CO, NO_x, NMOC), some of which react to form secondary pollutants (e.g. ozone (O_3), secondary organic aerosols (SOA), oxygenated NMOC). Large and frequent wildfires can significantly impact local and regional air quality and constitute a serious threat to human health (Bowman and Johnston 2005). An estimated average 339 000 deaths each year (260 000–600 000 deaths annually) are attributed to wildfire smoke exposure (Johnston *et al.* 2012). Studies have clearly and consistently demonstrated that wildfire smoke PM is associated with respiratory effects (Henderson *et al.* 2011), but the effects of other smoke components have not been well studied. For example, little is known about health effects with longer latencies, such as cancer, owing to few studies of how carcinogens in smoke, such as polycyclic aromatic hydrocarbons (PAHs), benzene and free radicals, affect exposed populations (Naeher *et al.* 2007). Despite this uncertainty, wildfire smoke is estimated to have a large health burden (Johnston *et al.* 2012) and a high cost to society (Kochi *et al.* 2010).

Fewer epidemiologic studies have evaluated the health impacts of air pollution resulting from wildfires compared with urban PM (Naeher *et al.* 2007; Dennekamp *et al.* 2015) in part because wildfires are short-lived and unpredictable (Henderson *et al.* 2011). A better understanding of the components and drivers of smoke emissions is needed to inform research on health effects of wildfire smoke. Such information would also inform decisions aimed at minimising exposure to toxic components of wildfire smoke. This is relevant for public health emergency responders and wildfire managers to address smoke exposures of communities (Kochi *et al.* 2010), but also relevant to wildfire managers to address the issue of occupational exposures on the ground.

The purpose of the present paper is to review the features of wildfire smoke that should be considered when assessing public health risk. These include combustion factors that influence smoke production (i.e. fuel characteristics and combustion conditions), external environmental factors that influence smoke plume dispersion (i.e. climate, meteorology and topography) and individual components of smoke that have the potential to impact on health. A better understanding on how these features influence public health risk will improve the development of evidence-based guidelines for public health response to wildfire smoke events. The *Factors that influence smoke production (emissions)*section briefly reviews how smoke is produced and the main factors that influence its production, and summarises the current scientific standing on emissions from major biomass-burning sources. In the *External environmental factors* section, we explore how environmental factors influence smoke production and how smoke is transported downwind and impacts on populations. The components of smoke pertinent to population exposures and public health risk are discussed in the *Main components of smoke pertinent to public health risk* section .

Factors that influence smoke production (emissions)

The impact of smoke on air quality and thus human health depends on the exposure to smoke, which is influenced by the amount and composition of emissions. Emissions are calculated as follows

$$E(i) = A \times F \times CC \times EF(i) \quad (1)$$

where E(i) is the emission for species (i), A is the area burned, F is the fuel load, CC is the combustion completeness and EF(i) is the emission factor for species (i).

The parameters in Eqn 1 are driven by various factors including fire and fuel characteristics, external environmental factors and the interactions of these with the combustion process (Fig. 1).

Emission factors

Characterisation of emission factors (EFs) helps quantify trace gases and particles released to the atmosphere. The EFs are also an essential input into smoke forecasting models to assess surface concentrations downwind and hence assess exposures to populated areas. Emission factors are also required for models to predict formation and exposure to secondary smoke pollutants, such as O_3.

EFs of selected species for major biomes have recently been reviewed and summarised by Akagi *et al.* (2011) and are shown

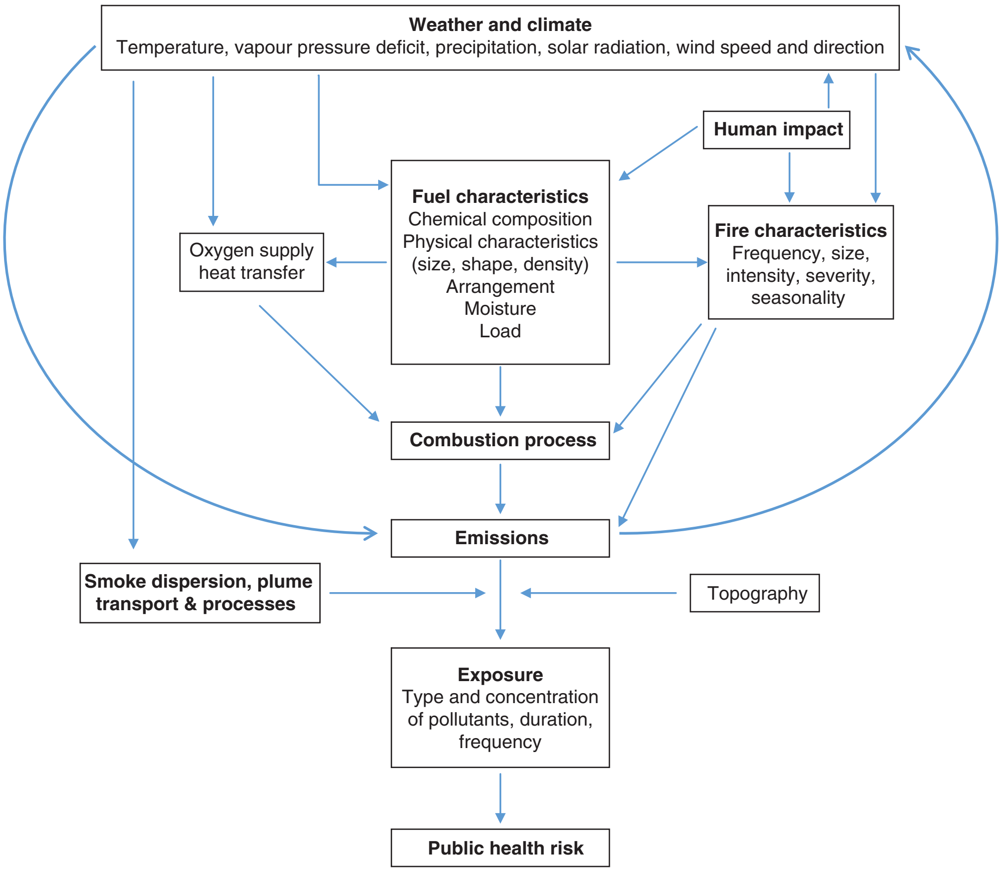

Fig. 1. Interactions between the various factors that influence smoke production and hence impact on public health.

Table 1. Emission factors (EF) (g kg^{-1}) for species emitted from different types of biomass burning

Adapted from Akagi *et al.* (2011). EF are shown with an estimate of the natural variation in parenthesis, when available

	Tropical forest	Savanna	Boreal forest	Temperate forest	Peatland
Carbon dioxide (CO_2)	1643 (58)	1686 (38)	1489 (121)	1637 (71)	1563 (65)
Carbon monoxide (CO)	93 (27)	63 (17)	127 (45)	89 (32)	182 (60)
Methane (CH_4)	5.07 (1.98)	1.94 (0.85)	5.96 (3.14)	3.92 (2.39)	11.8 (7.8)
Acetylene (C_2H_2)	0.44 (0.35)	0.24 (0.10)	0.18 (0.10)	0.29 (0.10)	0.14 (0.093)
Ethylene (C_2H_4)	1.06 (0.37)	0.82 (0.35)	1.42 (0.43)	1.12 (0.35)	1.79 (0.72)
Ethane (C_2H_6)	0.71 (0.28)	0.66 (0.41)	1.79 (1.14)	1.12 (0.67)	–
Propylene (C_3H_6)	0.64 (0.43)	0.79 (0.56)	1.13 (0.60)	0.95 (0.54)	2.3 (0.74)
Propane (C_3H_8)	0.126 (0.060)	0.10 (0.067)	0.44	0.26 (0.11)	–
1,3-Butadiene (C_4H_6)	0.039	0.052 (0.028)	0.14	–	
Isoprene (C_5H_8)	0.13 (0.056)	0.039 (0.027)	0.15	–	
n-Hexane (C_6H_{14})	0.01	0.013 (0.0074)	0.055	–	
Heptane (C_7H_{16})	5.60×10^{-3}	0.0070 (0.0072)	0.048	–	
Benzene (C_6H_6)	0.39 (0.16)	0.20 (0.084)	1.11	–	2.46 (1.21)
Toluene ($C_6H_5CH_3$)	0.26 (0.13)	0.080 (0.058)	0.48	–	1.21 (0.69)
Xylenes (C_8H_{10})	0.11 (0.082)	0.014 (0.024)	0.18	–	
Ethylbenzene (C_8H_{10})	0.050 (0.036)	0.006 (0.010)	0.051	–	
α-Pinene ($C_{10}H_{16}$)	–	–	1.64	–	
β-Pinene ($C_{10}H_{16}$)	–	–	1.45	–	
Ethanol (CH_3CH_2OH)	–	–	0.055	–	
Methanol (CH_3OH)	2.43 (0.80)	1.18 (0.41)	2.82 (1.62)	1.93 (1.38)	5.36 (3.27)
Phenol (C_6H_5OH)	0.45 (0.088)	0.52 (0.36)	2.96	0.33 (0.38)	4.36 (5.06)
Formaldehyde (HCHO)	1.73 (1.22)	0.73 (0.62)	1.86 (1.26)	2.27 (1.13)	1.69 (1.62)
Glycolaldehyde ($C_2H_4O_2$)	2.84	0.81 (0.38)	0.77	0.25 (0.45)	2.62 (4.18)
Acetaldehyde (CH_3CHO)	1.55 (0.75)	0.57 (0.30)	–	–	2.81 (1.36)
Acrolein (C_3H_4O)	0.65 (0.23)	–	–	–	
Furaldehydes	0.29 (0.0010)	–	–	–	
Acetone (C_3H_6O)	0.63 (0.17)	0.16 (0.13)	0.75	–	1.08 (0.29)
Methacrolein (C_4H_6O)	0.15 (0.045)	–	0.087	–	
Crotonaldehyde (C_4H_6O)	0.24 (0.068)	–	–	–	
Methyl ethyl ketone (C_4H_8O)	0.50 (0.21)	–	0.22	–	–
Furan (C_4H_4O)	0.41 (0.10)	0.17 (0.058)	0.80 (0.50)	0.20 (0.21)	1.51 (0.37)
Acetol ($C_3H_6O_2$)	1.13 (0.12)	0.45 (0.24)	–	–	1.92 (0.20)
Acetonitrile (CH_3CN)	0.41 (0.10)	0.11 (0.058)	0.61	–	3.70 (0.90)
Propenenitrile (C_3H_3N)	0.04 (0.01)	0.051 (0.022)	–	–	
Propanenitrile (C_3H_5N)	0.09	0.031 (0.014)	–	–	
Pyrrole (C_4H_5N)	0.12 (0.038)	–	–	–	
Formic acid (HCOOH)	0.79 (0.66)	0.21 (0.096)	0.57 (0.46)	0.35 (0.33)	0.54 (0.71)
Acetic acid (CH_3COOH)	3.05 (0.90)	3.55 (1.47)	4.41 (2.66)	1.97 (1.66)	7.08 (3.40)
Hydrogen cyanide (HCN)	0.42 (0.26)	0.41 (0.15)	1.52 (0.82)	0.73 (0.19)	5.0 (4.93)
Carbonyl sulfide (OCS)	0.025	–	0.46 (0.47)	–	1.20 (2.21)
Chloromethane(CH_3Cl)	0.053 (0.038)	0.055 (0.036)	0.059	–	
1,2-Dichloroethane ($C_2H_4Cl_2$)	–	–	1.29×10^{-3}	–	
Methyl iodide (CH_3I)	2.50×10^{-3} (3.45×10^{-3})	5.06×10^{-4} (3.88×10^{-4})	7.88×10^{-4}	–	
Dichlorodifluoromethane (CCl_2F_2)	$2.80 \times 10-3$	–	–	–	
Ethyl chloride (C_2H_5Cl)	–	–	–	7.47×10^{-4}	
Ammonia (NH_3)	1.33 (1.21)	0.52 (0.35)	2.72 (2.32)	0.78 (0.82)	10.8 (12.4)
Methyl nitrate (CH_3ONO_2)	8.29×10^{-3} (1.60×10^{-2})	5.1×10^{-4} (3.7×10^{-4})	2.83×10^{-3}	–	
Hydrogen (H_2)	3.36 (1.30)	1.70 (0.64)	–	2.03 (1.79)	–
Sulfur dioxide (SO_2)	0.40 (0.19)	0.48 (0.27)	–	–	–
Nitrous acid (HONO)	1.18	0.2	–	0.52 (0.15)	–
Nitrogen oxides (NO_x as NO)	2.55 (1.40)	3.9 (0.80)	0.90 (0.69)	2.51 (1.02)	0.80 (0.57)
Nitrous oxide (N_2O)	–	–	0.41	0.16 (0.21)	–
NMOC (identified)	26.0 (8.8)	12.4 (6.2)	29.3 (10.1)	11.9 (7.6)	48.7 (32.4)
NMOC (identified + unidentified)[A]	51.9	24.7	58.7	23.7	97.3
Total particulate carbon	5.24 (2.91)	3.00 (1.43)	–	–	–
$PM_{2.5}^{B}$	9.1 (3.5)	7.17 (3.42)	15.3 (5.9)	12.7 (7.5)	–
PM_{10}	18.5 (4.1)	–	–	–	
Black carbon (BC)	0.52 (0.28)	0.37 (0.20)	–	–	0.20 (0.11)
Organic carbon (OC)	4.71 (2.73)	2.62 (1.24)	–	–	6.23 (3.60)

[A] Estimated.

[B] PM_1–PM_5 categorised as $PM_{2.5}$.

in Table 1. Major biomes include tropical forests (5% of global land surface), savannas (22% of global land surface), boreal and temperate forests (6–10% of global land surface) and peatlands (3% of global land surface). The type of biomass burned is a major factor influencing emissions (Meyer *et al.* 2012) and EFs can be quite different between biomes. According to Table 1, fires in peatland, boreal and extratropical forests produce more CO, particulate matter with aerodynamic diameter less than 2.5 μm ($PM_{2.5}$), and other air toxics (including benzene, toluene, acetic acid, hydrogen cyanide and ammonia) per kilogram of fuel burned than fires in the savannas or tropical forests. This is partially driven by fire intensity. Fires in savannas and tropical forests are less intense, resulting in lower amounts of smoke produced compared with high-intensity crown fires observed in boreal and extratropical forests (Conard and Ivanova 1997; Volkova *et al.* 2014). Furthermore, as boreal forests contain large amounts of fuel, both above and below ground, smoke emissions can be significant. Although EFs for tropical forests are low, tropical ecosystems have the highest fuel loads, especially if the forest is grown on peat (Langmann *et al.* 2009), and therefore may emit significant amount of smoke. Peatlands possess high fuel loads, can burn over extended periods of time (Page *et al.* 2002), emit large amounts of reduced compounds (Christian *et al.* 2003) and therefore have a significant impact on air quality.

Although biomes can have a major influence on smoke production, emissions are also highly dependent on the combustion process and can vary during seasons owing to the alterations in water content and weather conditions (Korontzi *et al.* 2003).

Combustion process

The processes that take place in combustion of biomass have been described elsewhere (Yokelson *et al.* 1997; Koppmann *et al.* 2005; Sullivan and Ball 2012). Here, the focus is on identifying how combustion influences the products released from fires. Combustion of biomass proceeds through different stages (i.e. thermal degradation, oxidation of volatiles in flaming combustion, char oxidation in smouldering combustion), each with different chemical processes that result in different emissions (Andreae and Merlet 2001). Within a fire, these processes occur simultaneously and are interlinked.

When ignition begins, a complex mixture of tar and gas products is released, which, when diluted with air, forms a flammable mixture. When this mixture ignites, flaming combustion occurs, releasing carbon dioxide (CO_2) and water vapour (Koppmann *et al.* 2005). At low temperatures (typically <400 K), methanol, light aldehydes, and formic and acetic acid are the dominant emissions. Once over 450 K, $\sim$80% of the fuel material is thermally decomposed and large amounts of gaseous compounds are emitted (Koppmann *et al.* 2005). Combustion efficiency (CE), the fraction of fuel carbon emitted by the fire that is completely oxidised to CO_2, is an important parameter to characterise the combustion stage. When CE surpasses $\sim$90%, a fire is typically in the flaming phase, when C, H, N and S in the fuel are converted into highly oxidised gases such as CO_2, H_2O, NO_x and N_2O, and sulphur dioxide (SO_2) respectively (Andreae and Merlet 2001), and most of the black (or elemental) carbon particles are produced. A CE of less than $\sim$85% indicates a smouldering combustion, which is a slow, low-temperature, flameless form of combustion that produces most of the CO, methane (CH_4), NMOC, ammonia (NH_3), amines, nitriles and primary organic aerosol (Yokelson *et al.* 1996; Akagi *et al.* 2011).

Oxygen supply, temperature and fuel characteristics are important factors that govern the different stages of the combustion process and hence the type of compounds emitted into the air (Fig. 1). For example, the open structure of grass favours efficient heat transfer and rapid mixing of air, resulting in fierce, flaming combustion. Fine woody fuels, grass, litter and foliage are mainly consumed by flaming combustion, whereas large woody fuels (logs, stumps) require sufficient heat input to sustain a flaming combustion and therefore are mainly consumed by smouldering combustion. Combustion of ground fuels (peat, duff, organic soil) is also limited by air supply, resulting in a slow smouldering combustion. Smouldering combustion leads to increased emissions of reduced compounds, which can persist for days or even weeks and constitute a serious health hazard.

Variability in fuel moisture causes variability in the relative amounts of biomass consumed by flaming and smouldering combustion and hence on the amount and composition of emissions. Higher moisture content in the fuel produces water vapour that lowers the combustion temperature and hence favours smouldering (Goldammer *et al.* 2009), leading to increased emission of air pollutants.

External environmental factors

Weather and climate conditions

Weather and climate conditions affect the behaviour of wildfires by influencing the size, severity and frequency of fires, plume distribution and exposure times to populations.

High temperatures and other extreme weather conditions may increase the length of fire seasons and result in longer burning periods. Natural phenomena associated with sea-surface temperature anomalies and extended droughts can also influence the frequency and severity of wildfires (Wang *et al.* 2010*a*). For instance, El Niño–Southern Oscillation (ENSO) in the tropical, subtropical and some boreal regions is associated with more frequent droughts and intense fires, and has been associated with greater risks of human mortality attributable to smoke from wildfires (Johnston *et al.* 2012). The Indian Ocean Dipole (IOD) (Indonesia) or Indian Niño is also an irregular oscillation of sea-surface temperatures in which the western Indian Ocean becomes warmer and tends to cause droughts in adjacent land areas of Indonesia and Australia (Thompson *et al.* 2001). In 1997, IOD triggered intense fires over Indonesia, causing a regional O_3 increase.

In a warmer climate, higher temperatures and precipitation changes are expected to be the main climate drivers for fires by altering their frequency and intensity (Keywood *et al.* 2013). Wildfire activity is projected to increase under future atmospheric CO_2 concentration scenarious, possibly doubling the burned area by the end of the 21st century in areas such as Canada (Amiro *et al.* 2009) and Portugal (Carvalho *et al.* 2011). According to Spracklen *et al.* (2009), changes in the burned area are ecosystem-dependent, with the forests of the Pacific Northwest and Rocky Mountains experiencing the greatest increases of 78 and 175% respectively. An increase in burned area is projected to result in doubling of wildfire carbonaceous aerosol emissions by mid-century (Spracklen *et al.* 2009).

Factors that influence plume dispersion

Heat and moisture released by fires and meteorological conditions (e.g. atmospheric stability) are some of the factors that determine plume rise and plume dispersion, and hence impacts downwind (Heilman *et al.* 2014). Plume height is also strongly influenced by fire intensity and the fire-induced convection can rapidly transport emissions upward, with injection heights reaching up to 2–6 km (Colarco *et al.* 2004) resulting in effective transport of pollutants in the free troposphere over thousands of kilometres without chemical removal (de Gouw *et al.* 2006). Pollutants are transported from these elevated layers to the ground surface when plumes gradually subside and are entrained into the planetary boundary layer (Colarco *et al.* 2004), where they can undergo chemical transformations (de Gouw *et al.* 2006). Therefore, concentrations downwind can be significantly different to fresh emissions owing to dilution during transport when clean air is entrained in the smoke plume; chemical, photochemical and physical transformations during transport, and removal processes.

Weather patterns influence the movement of smoke plumes and dispersion of pollutants in the atmosphere via wind currents, vertical mixing and rainfall (Kinney 2008). Air pollution episodes can occur with atmospheric conditions that limit both vertical and horizontal dispersion (Kinney 2008). Also, during dry weather conditions, and for large injection heights, the aerosol particles can be transported by winds over thousands of kilometres before being removed from the atmosphere by wet deposition (Langmann *et al.* 2009). For example, the plumes from large wildfires in the tropics and boreal forests can be transported over thousands of kilometres (Watson *et al.* 1990; de Gouw *et al.* 2006), affecting the air quality not only in rural regions but also in regions with large anthropogenic emissions that are thousands of kilometres away from the fires (Wotawa and Trainer 2000; Jaffe *et al.* 2004; Cottle *et al.* 2014). Smoke plumes have been traced with satellite data, with distances exceeding 1600 km from wildfires (Scala *et al.* 2002).

Long-range episodic transport of trace gases and particles from fires into populated areas has been observed across the world. In 2010, an unprecedented intensive heat wave provoked thousands of wildfires during summer, impacting severely on the air quality in Moscow, where concentrations of CO and PM_{10} exceeded 10 mg m^{-3} and 700 μg m^{-3} respectively (Konovalov *et al.* 2011). The changes in air quality were caused by transport of smoke from intensive fires in the north of Ryazan, a city situated ~180 km south-east of Moscow. Significant enhancement in O_3, CO and aerosol concentrations were also observed in Alaska, Canada and the north-western US as a result of long-range transport of Siberian fire plumes (Jaffe *et al.* 2004; Cottle *et al.* 2014). Another example was reported in 2002 when a dramatic increase in wildfire activity in the province of Quebec, Canada, produced a PM episode that reached Baltimore, Maryland, located more than 1500 km from the source. In this episode, $PM_{2.5}$ concentrations reached 86 μg m^{-3}, exceeding the 24-h national ambient air quality standard (Sapkota *et al.* 2005). Another study in Quebec found that concentrations of $PM_{2.5}$, mercury (Hg) and carbon oxides increased after a wildfire, reducing the visibility in Quebec City, Montreal and Ottawa, as well as other rural areas located more than 250 km away from the fire (Wang *et al.* 2010*b*). During the uncontrolled forest and peat fires in Indonesia in 2013, smoke was transported 200–500 km to Singapore, creating hazardous air-quality conditions (Betha *et al.* 2014).

Elevation and topography can affect smoke accumulation and plume distribution. For example, when mountain slopes are warmed by the sun, air is heated and can move upslope, bringing smoke and fire with it. When the terrain cools (e.g. at night), the air begins to descend, creating a different airflow that can alter the smoke dispersal pattern during the day (Robock 1991). In valleys and small basins, temperature inversions are common. The cooler air near the ground prevents upward air movement, which results in smoke accumulation for several weeks under the inversion layer (Achtemeier 2005). These smoke accumulations can reduce roadway visibility and cause adverse health effects in nearby populated areas as a result of longer exposure times.

Modelling and forecasting tools

One of the main challenges in smoke management is predicting the magnitude and location of smoke effects. Thus, new studies are developing a variety of tools in order to predict smoke effects after a wildfire. Models for predicting smoke effects of forest fires consist of four main components: (1) emissions source; (2) plume rise; (3) plume dispersion; and (4) chemical and physical transformations within the plume (Goodrick *et al.* 2013; Yao *et al.* 2013).

The main challenges to develop tools for forecasting wildfires are to predict plume distribution; some of the initial models have failed because they have mistakenly assumed that smoke travels in a straight line under homogeneous conditions, and have failed to consider other important factors such as wind, topography and meteorological conditions, which vary in space and time and influence plume trajectory (Goodrick *et al.* 2013). The newest approaches are based on modelling frameworks that can be adapted to regional conditions and specific situations by enabling simulations of cumulative smoke impacts from fires in forest and agricultural areas (Goodrick *et al.* 2013). These approaches allow plume structure to embody a range of different behaviours characteristic of forests and the myriad plume structures to ultimately improve the predictive power of the model (Goodrick *et al.* 2013).

In summary, fuel characteristics, combustion conditions, and weather and climate are major factors that should be considered when assessing public health risk due to smoke exposure. Based solely on emissions of toxic smoke components, fires in peatland and boreal forests are likely to pose the greatest health risk. For peat fires, this is due to the high fuel load and slow smouldering combustion releasing mainly reduced compounds such as CO, NMOC and PM (primarily composed of organic carbon) for extended periods of time. Such extreme air pollution events occurred in south-east Asia in 1997–98 and 2013 from uncontrolled fires in peatland swamp forests, causing dangerous levels of PM and other toxic components in Indonesia and

Table 2. Major health-damaging pollutants from biomass combustion; recommended concentrations by the WHO and the Ambient Air Quality Criteria (AAQC) are indicated for each component

Source: Naeher *et al.* 2007; WHO 2006; Ontario Ministry of the Environment 2012

Compound	Examples	Concentration allowed by WHO and AAQCs	Mode of toxicity
Particulate matter (PM)	Fine particles ($PM_{2.5}$)	25 $\mu g\ m^{-3}$ (24-h)	Can cause or aggravate cardiovascular and lung diseases, heart attacks and arrhythmias, affect the central nervous system and the reproductive system and cause cancer. The outcome can be premature death.
Inorganic gases	Carbon monoxide (CO)	30 $mg\ m^{-3}$ (1-h)	Can lead to heart disease and damage to the nervous system and cause headaches, dizziness and fatigue.
	Ozone (O_3)	120 $\mu g\ m^{-3}$ (8-h)	Can decrease lung function, aggravate asthma and other lung diseases. Can lead to premature mortality.
	Sulfur dioxide (SO_2)	20 $\mu g\ m^{-3}$ (24-h)	Aggravates asthma and can reduce lung function and inflame the respiratory tract. Can cause headache, general discomfort and anxiety.
	Nitrogen dioxide (NO_2)	200 $\mu g\ m^{-3}$ (1-h)	Can affect the liver, lung, spleen and blood. Can aggravate lung diseases, leading to respiratory symptoms and increased susceptibility to respiratory infection.
Hydrocarbons	Unsaturated, e.g. 1,3-butadiene	10 $\mu g\ m^{-3}$ (24-h)	Irritant, carcinogenic, mutagenic
	Saturated, e.g. *n*-hexane	2500 $\mu g\ m^{-3}$ (24-h)	Irritant, neurotoxic
	Polycyclic aromatic hydrocarbons (PAH)	5×10^{-5} $\mu g\ m^{-3}$ (24-h)	Carcinogenic. Other effects may be irritation of the eyes, nose, throat and bronchial tubes.
	Benzene	2.3 $\mu g\ m^{-3}$ (24-h)	A human carcinogen, which can cause leukaemia and birth defects. Can affect the central nervous system and normal blood production, and can harm the immune system.
	Styrene	400 $\mu g\ m^{-3}$ (24-h)	Carcinogenic, mutagenic
Oxygenated organics	Aldehydes, e.g.		Irritant, carcinogenic, mutagenic
	Acrolein	0.4 $\mu g\ m^{-3}$ (24-h)	
	Formaldehyde	65 $\mu g\ m^{-3}$ (24-h)	
	Organic alcohols and acids, e.g.		Irritant, teratogenic
	Methanol	4000 $\mu g\ m^{-3}$ (24-h)	
	Acetic acid	2500 $\mu g\ m^{-3}$ (24-h)	
	Phenols, e.g. cresol	75 $\mu g\ m^{-3}$ (24-h)	Irritant, carcinogenic, mutagenic, teratogenic
	Quinones	15 $\mu g\ m^{-3}$ (24-h)	Irritant, allergenic, causes oxidative stress and inflammation, redox-active, possibly carcinogenic
Chlorinated organics	Methylene chloride	220 $\mu g\ m^{-3}$ (24-h)	Central nervous system depressant (methylene chloride), possible carcinogens
	Methyl chloride	320 $\mu g\ m^{-3}$ (24-h)	
Mercury		1 $\mu g\ m^{-3}$ (annual)	Toxic effects on the nervous, digestive and immune systems, and on lungs, kidneys, skin and eyes
Free radicals	Semiquinone-type radicals		Redox-active, causes oxidative stress and inflammatory response, possibly carcinogenic

neighbouring countries (Heil and Goldammer 2001; Betha *et al.* 2014). Similar but less intense events were observed from fires in boreal and temperate forests (DeBell *et al.* 2004; Sapkota *et al.* 2005; Wu *et al.* 2006; Hodzic *et al.* 2007; Reisen *et al.* 2011*a*). Although tropical forests have lower EFs, they have higher fuel loads, and fires in those regions may contribute significantly to emissions and hence impacts on populations.

The major components likely to cause a public health risk owing to their toxicity, irritancy and carcinogenicity include PM, CO, NO_x, NH_3, SO_2, and certain NMOC (e.g. formaldehyde, phenol, benzene) (Naeher *et al.* 2007). Based on health studies conducted in polluted regions, mainly urban cities, the World Health Organization has established standard levels for different air pollutants (Table 2). Exceedances of these levels may result in deleterious health effects. Intensity, duration and frequency of exposure determine whether adverse health effects are likely. These are further discussed in the *Main components of smoke pertinent to public health risk* section.

Main components of smoke pertinent to public health risk

This section provides a review on the components of smoke that are pertinent to human health. It looks at the characteristics of the smoke components and reviews the current information available to assess human health risk. The components include particles, CO, O_3, other inorganic trace gases, NMOC, Hg and pollutant mixes. The health effects and air quality guidelines for the pollutants are shown in Table 2.

Particulate matter

Particulate matter (solid or liquid particles suspended in air) is directly emitted from fires and formed through secondary processes that may involve NMOCs (Akagi *et al.* 2011). Less than 5% of carbon is released as PM, which is mainly composed of organic and black carbon and small contributions from inorganic species (e.g. nitrate, sulphate, ammonium and chloride) (Reid *et al.* 2005; Capes *et al.* 2008; Vakkari *et al.* 2014).

Table 3. Particle concentrations measured during major wildfire events

Location	Max hourly PM ($\mu g\ m^{-3}$)	Max daily PM ($\mu g\ m^{-3}$)	Duration	Reference
Southern California (2003)	769 (PM_{10})	~400 (PM_{10})	7 days	Phuleria *et al.* (2005)
British Columbia (2003)		200 ($PM_{2.5}$) 250 (PM_{10})	>30 days	Moore *et al.* (2006)
Portugal (2003)	300–500 (PM_{10})		June–September	Pio *et al.* (2008)
Indonesia (1997)		1800 (PM_{10})	>2 months	Kunii *et al.* (2002)
Singapore (2013)		329 ($PM_{2.5}$)	9 days	Betha *et al.* (2014)
Victorian Alpine fires (2006–07), Ovens, Vic. (Australia)	1800 ($PM_{2.5}$)	1100 ($PM_{2.5}$)	69 days	Reisen *et al.* (2011)
Melbourne (Australia)		~120 ($PM_{2.5}$)		

Biomass burning produces mainly fine particles less than 2.5 μm in diameter ($PM_{2.5}$), with a peak in the size distribution between 0.15 and 0.3 μm (Reid *et al.* 2005). The differences in the particle size and composition depend on the type of fuels burned and combustion phase. Smaller particle sizes have been observed for fires in tropical forests compared with fires in boreal forests (Eck *et al.* 2003). Flaming combustion tends to produce smaller particles rich in elemental carbon. During smouldering combustion, particles tend to be formed by condensation of volatilised organics onto existing particles and surfaces, producing particles rich in organic carbon (~80% of the mass) (Reid and Hobbs 1998; Reid *et al.* 2005). Similarly, in crown fires with high fire intensity, transport of oxygen into the interior flame zone is reduced, thereby producing a less efficient combustion and increasing particle size and production (Reid and Hobbs 1998). Peat fires also tend to produce larger and more hygroscopic particles (Eck *et al.* 2003). During transport of smoke plumes, both coagulation and condensation occur, leading to an increase in particle size with increasing time and distance from the original source (Eck *et al.* 2003). PM still remains small enough to remain in the air for days or weeks (Seinfeld and Pandis 2006) and can be transported over large distances, in particular for large injection heights.

Particle size, shape and composition are important factors to consider for health effects. Small particles, particularly $PM_{2.5}$, can penetrate into the deeper parts of the lungs where removal processes are slower and less efficient. Ultrafine particles (diameters smaller than 0.1 μm) are also able to penetrate the lung lining and enter the blood, potentially causing harmful effects (Highwood and Kinnersley 2006). There has also been an increased focus on the chemical composition of particles to assess if any particular components of a particle are responsible for the adverse health impacts (Harrison and Yin 2000). Carbonaceous material can make up a substantial proportion of the total PM mass and may have potential implications for human health (Mauderly and Chow 2008; Janssen *et al.* 2011; Rohr and Wyzga 2012). Owing to their shape, carbonaceous particles offer a large surface area to initiate production of free radicals and therefore have a greater potential to cause inflammation (Highwood and Kinnersley 2006). Furthermore, the chemical components in wildfire PM have been shown to be more toxic to the lung than ambient PM (Wegesser *et al.* 2009). These findings are consistent with a recent study that showed significantly higher excessive lifetime carcinogenic risk from exposure to biomass-burning aerosols compared with the risk from exposure to urban air aerosols during a haze event in Singapore (Betha *et al.* 2014).

Studies have shown that fine particles consistently exceed air-quality guidelines during wildfire events. Table 3 shows particle concentrations measured during major wildfire events in the US, Europe, Asia and Australia. The results clearly show the significant impact of particle pollution on communities downwind from the fires. Some of the particle pollution events lasted from days to weeks.

$PM_{2.5}$ from wildfire smoke has been strongly associated with respiratory effects (exacerbations of obstructive lung disease, bronchitis and pneumonia) in numerous epidemiologic studies (Henderson *et al.* 2011; Youssouf *et al.* 2014; Liu *et al.* 2015). Associations with other health outcomes (e.g. mortality, cardiovascular disease, lower birth weight) have been demonstrated, but with less consistency (Liu *et al.* 2015). Long-term mortality risk (15%) has been associated with PM levels of 35 and 70 $\mu g\ m^{-3}$ for $PM_{2.5}$ and PM_{10} respectively. Thus, an average annual concentration of 10 $\mu g\ m^{-3}$ has been chosen as an air-quality guideline for long-term exposures to $PM_{2.5}$ (WHO 2006). For short-term exposures, a 24-h mean air quality guideline of 25 $\mu g\ m^{-3}$ has been established in order to protect against peaks of pollution that can lead to increased mortality (Table 2).

Carbon monoxide

CO is an odourless, colourless and tasteless inorganic gas produced more abundantly when incomplete combustion occurs. Based on the EFs from different types of biomass burning (Table 1), peatland fires produce the most CO, followed by boreal forests. CO has a residence time in the troposphere of several months; thus, CO from biomass burning can be distributed over great distances and can be used as a tracer of biomass-burning plumes in the troposphere (McMillan *et al*. 2003).

Intense fires in peat swamp forests in Indonesia in 1997 increased CO concentrations approximately two orders of magnitude above background levels (Heil and Goldammer 2001). The smoke plumes reached into the upper troposphere and were transported towards the southern subtropics, causing anomalous increases in CO concentrations thousands of kilometres away (Matsueda *et al.* 1998; Fujiwara *et al.* 1999). Fires in boreal forests were also a significant contributor to CO emissions into the mid-latitudes and a dominant source of CO concentrations in regional areas of the eastern and south-eastern US (Wotawa and Trainer 2000).

Increased concentrations of CO have been observed in smoke plumes and significantly enhanced CO concentrations above background levels were measured in downwind locations (Brunke *et al.* 2001; DeBell *et al.* 2004; Bertschi and Jaffe 2005; Dutkiewicz *et al.* 2011). However, the increases in CO were far less than those observed for PM levels. Measured ground concentrations remained generally within air-quality guidelines, with a few studies reporting enhancements of CO concentrations above air-quality guidelines as a result of impacts from fire plumes (Aditama 2000; Kunii *et al.* 2002; Konovalov *et al.* 2011).

In general, concentrations of CO related to wildfire smoke have not been shown to pose a significant hazard to communities. However, elevated concentrations of CO have been measured on the fire line, where firefighters can be exposed to hazardous concentrations of CO (Reinhardt and Ottmar 2004; Reisen and Brown 2009; Miranda *et al.* 2010; Reisen *et al.* 2011*b*). Under those circumstances, exposure to CO can cause headaches, dizziness, disorientation and visual impairment (WHO 1999; Raub *et al.* 2000). Individuals with pre-existing cardiovascular problems may experience chest pain and cardiac arrhythmias.

Ozone

The production of O_3 from wildfires has been recently reviewed by Jaffe and Wigder (2012) and is briefly summarised here. O_3 is formed in the troposphere, mainly by reaction of NO_2 and hydrocarbons through NMOCs–NO_x photochemistry. Therefore, O_3 concentrations are dependent on chemical and photochemical reactions and hence on concentrations of precursor NMOCs and NO_x. NO_x is primarily produced from fuel nitrogen, which can vary considerably (Yokelson *et al.* 2008; McMeeking *et al.* 2009), but NO_x emissions are also dependent on combustion efficiency as described above. NO_x is typically the limiting O_3 precursor in smoke plumes (Jaffe and Wigder 2012). The typical lifetime of NO_x is less than 1 day. NO_x can be converted to HNO_3, aerosol NO_3 and peroxynitrates including peroxyacetyl nitrate (PAN) (Alvarado *et al.* 2010). Conversion of NO_x to PAN is a limiting factor on local *in situ* O_3 production, but sequestration of NO_x in the form of PAN can lead to increased O_3 concentrations further downwind in plumes aged more than 5 days by long-range transport and thermal decomposition of PAN (Alvarado *et al.* 2010; Parrington *et al.* 2013). The sequestration of NO_x in the form of PAN is more likely observed at colder temperatures of high latitudes (Alvarado *et al.* 2010) and can lead to continuing O_3 formation in well-aged plumes (Val Martin *et al.* 2006).

Enhancement of O_3 concentrations in fire plumes has been observed in several studies (Junquera *et al.* 2005; Lapina *et al.* 2006; Morris *et al.* 2006; Val Martin *et al.* 2006; Oltmans *et al.* 2010; Bossioli *et al.* 2012; Kang *et al.* 2014). Long-range transport of smoke from Siberian fires has been estimated to account for a 15-ppb increase in O_3 concentrations measured at a location in Washington State. This increase resulted in a maximum 8-h ground concentration of 96 ppb, thereby exceeding the 8-h standard of 80 ppb (Jaffe *et al.* 2004). Similarly large fires in Quebec in 2002 resulted in O_3 enhancements of $\sim$14 to 60 ppb above low mixing ratios of 30–35 ppb (DeBell *et al.* 2004). Highest enhancements were observed in well-aged plumes ($>$5 days) (Parrington *et al.* 2013).

External air masses may also influence the observed enhancements in O_3 concentrations. Junquera *et al.* (2005) showed that plumes over urban areas with anthropogenic NO_x sources can have significantly different photochemical impacts than plumes that do not encounter additional NO_x sources. O_3 can also be formed through photochemical reactions when smoke is trapped for several days in valleys or basins (Achtemeier 2005; Reisen *et al.* 2011*a*).

Conversely, photochemical formation of O_3 can be limited by high aerosol loading in the smoke plume, which reduces transmission of solar radiation (Duncan *et al.* 2003; Phuleria *et al.* 2005; *Real* et al. 2007; Verma *et al.* 2009). During the southern California wildfires in 2003, Phuleria *et al.* (2005) observed a decrease in hourly O_3 concentrations from 29 to 15 ppb, which they associated with limited photochemistry due to the thick smoke.

Because O_3 formation increases with more sunlight and higher temperatures, O_3 often reaches highest concentrations primarily during the warm half of the year (Kinney 2008). In the tropical and subtropical regions during dry seasons, O_3 concentrations have been observed in smoke plumes approximately three times higher than standard conditions of a fire-free atmosphere (Crutzen and Andreae 1990). Recirculation over the Indonesian maritime continent allows smoke to accumulate and tropical tropospheric O_3 to increase in periods without smoke (Keywood *et al.* 2013).

Concentrations of O_3 close to fires are not expected to be high enough to be of concern. However, increases of O_3 above the recommended levels due to wildfire emissions can occur in places where concentrations are already high owing to anthropogenic activities (Morris *et al.* 2006; Bein *et al.* 2008). In these areas, health effects such as reductions in lung function, lung inflammation and fatigue can be found at the population level even in young and healthy individuals (WHO 2006). Time-series studies on effects of air pollution have revealed positive, although small, associations between daily mortality and O_3 concentrations in North America and Europe (WHO 2006). There is considerable variation in response to O_3; based on current evidence, health effects can appear at concentrations greater than 120 $\mu g\ m^{-3}$ (Table 2).

Other inorganic gases

Inorganic trace gases are released in different phases of combustion. In general, NO_x, molecular N_2 and SO_2 are released during flaming combustion, whereas NH_3, amines and nitriles are associated with smouldering combustion (Andreae and Merlet 2001). NO_x emissions primarily depend on the N content in the fuel, and they play a significant role in tropospheric chemistry of NMOCs–NO_x–O_3. SO_x emissions are generally produced in negligible amounts owing to the low sulphur content in forest fuels. Higher S content is, however, observed in peatlands and areas rich in sulphur owing to accumulated volcanic sulphur through wet and dry deposition. Therefore, fires in those regions may cause significant increase in NO_2 and SO_2 emissions at close proximity. As an example, the 1997 extensive fires in Indonesia caused a major pollution event with levels of PM, CO, NO_x and SO_2 exceeding air quality guidelines in Indonesia (Aditama 2000), but not in Malaysia (Awang *et al.* 2000), Singapore (Emmanuel 2000) and Thailand (Phonboon *et al.* 1999) owing to lower concentrations of inorganic gases in the smoke plumes. Therefore, although inorganic gases are emitted from forest fires in varying amounts (Ward 1999), their concentrations in smoke plumes are only likely to pose a public health risk under extreme circumstances.

Non-methane organic compounds (NMOCs)

NMOCs are by-products of incomplete combustion directly emitted into the atmosphere from biomass-burning or products of oxidation of parent NMOCs. NMOCs include non-methane hydrocarbons (NMHCs) such as alkanes, alkenes, alkynes, aromatics and oxygenated NMHCs (e.g. alcohols, aldehydes and organic acids) (Urbanski *et al.* 2009). Similarly to CO, peatland fires have the highest emission factors of NMOCs, followed by boreal forests and tropical forests. Savannas and temperate forest produce the lowest amount of NMOCs. There is the potential for large variability in EFs within a biome due to differences in burning conditions (de Gouw *et al.* 2006; Lewis *et al.* 2013).

NMOCs have short atmospheric lifetimes (from hours to months) (Atkinson and Arey 2003) and play an important role in chemical and photochemical reactions to produce SOA and O_3 (de Gouw *et al.* 2006). NMOCs are disseminated from their sources over regional areas and can be transported globally. Owing to the variability in emissions, the removal processes by either OH reaction or scavenging by precipitation and the secondary production of NMOCs, understanding of how NMOCs from biomass-burning behave in the atmosphere is still developing (de Gouw *et al.* 2006; Yokelson *et al.* 2013).

Only a few studies have looked at the impact of smoke plumes on downwind surface NMOC concentrations (Ward *et al.* 2005; Blake *et al.* 2009; Evtyugina *et al.* 2013). Overall, although strong enhancements of NMOCs have been observed in smoke plumes (Sinha *et al.* 2003; de Gouw *et al.* 2006; Singh *et al.* 2010; Simpson *et al.* 2011; Akagi *et al.* 2013), the increase did not result in exceedances of air-quality guidelines. The low concentrations observed in downwind plumes are likely due to the chemical and photochemical reactions of the NMOCs and scavenging by precipitation. For example, emitted PAHs react in the atmosphere via reaction with OH radicals and photodegradation, which may explain their absence in daytime samples collected in Malaysia during the 1997 south-east Asia haze episode (Abas *et al.* 2004). However, a significant increase in PAH concentrations was observed in Indonesia, where concentrations of four-ring PAHs in affected areas were 40 to 60 times higher than those in unaffected areas (Kunii *et al.* 2002). Total PAH concentrations were ~200 ng m^{-3} and maximum benzo(a) pyrene concentration was 15 ng m^{-3}, exceeding the recommended annual guideline of 0.3 ng m^{-3}. Similarly high PAH concentrations (34 ng m^{-3}) were measured during peat fires in Brunei (Muraleedharan *et al.* 2000; Radojevic 2003). Other NMOCs such as benzene are overwhelmed by emissions from fossil fuel sources, in particular in urbanised areas of the northern hemisphere (Ward *et al.* 2005); however, biomass-burning is a major contributor to benzene in most parts of the southern hemisphere where biomass-burning sources dominate (Lewis *et al.* 2013).

Potential health effects from NMOCs, including PAHs, are inferred from knowledge of some of their components. Among the NMOCs, some of the few classes established for regulatory purposes are hydrocarbons, oxygenated organics and chlorinated organics (Table 2). Some NMOCs, such as benzene, formaldehyde and benzo(a)pyrene, are known carcinogens (IARC 2012), whereas other NMOCs (e.g. phenol, acetic acid, acrolein) are irritants for skin, eye, nose and throat. Some studies also suggest a link between benzene exposure and childhood cancer (Filippini *et al.* 2015; Vinceti *et al.* 2012).

On the fire line, exposures to benzene and formaldehyde can be significant and exceed occupational exposure standards (Reinhardt and Ottmar 2004; Reisen and Brown 2009). Similarly elevated hourly peak concentrations of benzene have been observed in smoke plumes (Blake *et al.* 2009; Evtyugina *et al.* 2013); however, concentrations generally do not approach air-quality thresholds. Although formaldehyde presents a health risk on the fire line, formaldehyde concentrations are unlikely to pose a health risk in downwind plumes owing to its short atmospheric lifetime (few hours).

Mercury

Mercury (Hg) can be a serious contaminant of ecosystems with subsequent consequences to human health. Vegetation accumulates Hg via atmospheric deposition, foliar accumulation of depositions, or uptake from roots when close to a natural or anthropogenic source of Hg (Pirrone and Mahaffey 2005). Hg concentration within plants varies and is highest in leaves, bark and roots.

The three atmospheric Hg species that contribute to inorganic Hg accumulation in vegetation are gaseous elemental mercury (Hg^0), reactive gaseous mercury (Hg^{II}) and particle-bound mercury (Hg(p)) (Schroeder and Munthe 1998). The concentrations of Hg^{II} and Hg(p) may be more substantially influenced by wildfires compared with Hg^0 (Pirrone and Mahaffey 2005). Hg emissions from wildfires also include resuspension of industrial Hg emissions that were deposited on trees and the soil within forests (Pirrone and Mahaffey 2005). Hg emitted from burning biomass reaches an elevated percentage (97–99%), i.e. there is almost complete transfer of Hg from vegetation to atmosphere during combustion (Pirrone and Mahaffey 2005). Gustin and Lindberg (2005) report that wildfires produce an estimated 200–1000 mg Hg per year globally.

Emissions of Hg are dominated by gaseous elemental Hg, although ~15% of Hg emitted from wildfire is in particulate form (Zhang *et al.* 2013). Elemental Hg has a lifetime of ~1 year before redeposition, whereas particulate Hg lasts from days to weeks (Turetsky *et al.* 2006). Emissions of Hg from small and large fires in boreal forests ranged from 0.4 to 116.8 t, with an average of 22.8 ± 7.5 t per year (Friedli *et al.* 2003). Fire emissions from boreal peatlands ranged from 1.5 to 7.0 mg Hg m^{-2}. Including estimation from upland and peatland soils indicated that ~340 t of Hg per year could be emitted across the circumboreal region (Turetsky *et al.* 2006).

Almost nothing is known about the health effects of Hg release from wildfires. Current knowledge of health effects from Hg are derived from studies of populations living close to mining areas or power plants. Based on these studies, the recommended concentration for Hg in air is 1 μg m^{-3} per year (WHO 2006). The inhalation of Hg vapour can produce harmful effects on the nervous, digestive and immune systems, lungs and kidneys, with increased risks of cardiovascular disease and severe neurological damage to humans (Diez 2009).

Free radicals

Wood smoke and combustion of forest fuels have been identified as a source of free radicals (Lachocki *et al.* 1989; Dost 1991; Ward 1999; Leonard *et al.* 2000; Leonard *et al.* 2007). Owing to their great reactivity, free radicals pose a health risk if they persist in the atmosphere and come in contact with human tissues. Currently, only limited information is available on the type and quantity of free radicals emitted during biomass burning, the persistence of free radicals in the atmosphere and the subsequent health effects. A study by Leonard *et al.* (2000) has shown that wood smoke is able to produce carbon-centred and reactive OH radicals and can in turn cause cellular and DNA damage. The study also showed that wildfire smoke contains both carbon radicals and precursors that are able to react and generate OH radicals as well as reactive oxygen species (ROS) after exposure to cells (Leonard *et al.* 2007). Ultrafine particles have shown the highest production of ROS, which is of particular concern as these particles can penetrate deep in the alveolar region of the lungs and may be absorbed into the bloodstream. Free-radical production increased with greater particle surface area (Leonard *et al.* 2007). The research also indicates that low temperature and incomplete combustion result in carbon radicals staying intact, whereas higher temperatures are likely to dissipate the carbon radicals and produce more potentially reactive OH radical precursors. This may suggest that high-intensity, flaming fires would pose a greater health risk than low-intensity, smouldering fires.

Pollutant mixes

Changes in plume composition with ageing

Smoke plumes are a complex mix of pollutants including CO_2, CO, CH_4, NMOC, aerosols, NO_x and other trace gases that can be subject to physical, chemical and photochemical processes during transport, leading to their transformation or removal. The primary controlling factors for these transformations include reactions with the hydroxyl radical ($OH^\bullet$), photolysis and scavenging by precipitation. The formation and removal processes are dependent on external factors including humidity, solar radiation, temperature, cloud cover, time of day and presence of other pollutants (de Gouw *et al.* 2006; Akagi *et al.* 2013). For example, NMHC such as C2–C4 alkenes and alkanes are typically lost over a few hours or few days respectively through NMOCs–NO_x–OH reactions in the continental boundary layer (Atkinson and Arey 2003). However, lower OH concentrations due to low humidity, low photolysis rates and high levels of CO and other pollutants in the fire plumes can reduce the removal of aromatic NMOCs (de Gouw *et al.* 2006). The photochemical processing initiated by reaction with OH leads to oxidation of NMOCs, with the subsequent formation of secondary pollutants such as O_3 and SOA (Urbanski *et al.* 2009; Vakkari *et al.* 2014; Keywood *et al.* 2015). Compounds formed by condensation of NMOC with multiple functional groups emitted during burning appear to be a major component of SOA (Keywood *et al.* 2013). The formation of SOA can be quite rapid and wildfire aerosol mass has been observed to increase by a factor of 1.5–2 over a period of a few days (Reid *et al.* 1998; Reid *et al.* 2005). This mixture is highly reactive as demonstrated by the relatively short atmospheric lifetimes of many of these compounds with respect to gas-phase reaction or photolysis (Keywood *et al.* 2013) and has a significant potential to influence tropospheric chemistry. Water-soluble NMOCs such as acetonitrile, methanol, acrolein and acetaldehyde can also be removed by precipitation.

Interaction between smoke plumes and other sources

Wildfire smoke plumes can also mix with materials from other sources, thereby strengthening their impact on air quality. For instance, over and downwind of Africa, aged biomass-burning aerosol plumes are often mixed with desert dust (Johnson *et al.* 2008). Similarly, synergistic effects on O_3 enhancements were observed when fire plumes dispersed over regions with high biogenic emissions (Bossioli *et al.* 2012).

The combined effects of urban pollution with wildfire smoke should also be considered because pollutant concentrations may change when wildfire smoke reaches urban areas. For example, in Mexico City, emissions from wildfires in the pine–savannas from the surrounding mountains near the city increased the concentrations of NO_x and hydrogen cyanide twice as much as the normally observed concentrations from forest-burning alone (Yokelson *et al.* 2007). The nitrogen enrichment in the fire emissions may be due to deposition of nitrogen-containing pollutants in the outflow from the urban area. This effect can possibly also be occurring in other parts of the world, when biomass-burning coexists with large urban areas (e.g. the tropics, south-eastern US, Los Angeles Basin) (Yokelson *et al.* 2007).

Effects of fire smoke can also be exacerbated in highly polluted cities. In 2004, extensive areas in Alaska (2.7 million ha) and Canada (3.1 million ha) were burned, resulting in a huge plume of smoke that eventually settled in Houston (Morris *et al.* 2006). Houston, a highly polluted city in the southern USA, is routinely affected by above-average O_3 levels due to numerous petrochemical production plants. O_3 concentrations increased 50 to 100% owing to the combined effects of urban pollution and wildfire smoke, resulting in the highest 8-h maximum O_3 levels ever recorded for a July day in a 5-year period that year (Morris *et al.* 2006). Similar effects were observed in Pittsburgh, a city burdened by large anthropogenic emissions, where smoke from the large 2002 Quebec wildfires resulted in exceedance of the 8-h O_3 National Ambient Air Quality standard (Bein *et al.* 2008).

Although the impacts of pollutant mixes on human health are not quite yet understood, several compounds produced by photochemical oxidation, such as aldehydes, are of interest owing to their potential health impact. Aldehydes are extremely irritating to the mucous membranes of the human body; formaldehyde and acetaldehyde are known and probable human carcinogens respectively (IARC 1999; IARC 2012). Other components within the pollutant mixes such as NMOCs may cause adverse health effects, because some of these components are highly irritant and probable human carcinogens. Nonetheless, the synergistic or additive effects of these components from the mixture of pollutants are still unknown (Urbanski *et al.* 2009).

Conclusions and research gaps

Wildfire smoke is a complex mixture of pollutants that are emitted into the atmosphere, transported downwind and impact on communities. Factors that drive emissions and exposures are important when assessing public health risk, as they provide crucial information to determine the type and concentration of pollutants that people are exposed to as well as the duration and frequency of exposure. The main points to be considered for public health response include the main components present in smoke, duration of exposure and factors influencing plume distribution (e.g. meteorology and topography).

The amount and type of emissions are strongly influenced by fuel characteristics and the physical and chemical processes during combustion (i.e. flaming or smouldering conditions).

- Emission factors are critical to determine impacts of wildfires, and therefore characterising fires in terms of each biome is useful to determine the potential impacts of biomass-burning in different regions and estimate the types and amount of toxic compounds released by wildfires. However, they are mostly presented as biome-averaged EFs that do not take into account the spatial and temporal complexities of the fuel characteristics and combustion processes. Owing to the spatial and temporal variability of each single factor in Eqn 1, the development of emission inventories is challenging (Larkin *et al.* 2014).
- Understanding combustion efficiency will also provide a better assessment of emissions of toxic compounds and their impact on health. During smouldering combustion, there is a higher conversion of fuel components to toxic compounds such as CO, NMOC and organic aerosols than during flaming combustion. Furthermore, smouldering combustion can be sustained over long periods, affecting communities close by.
- Emission of trace gases and particles into the atmosphere is also dependent on the fuel load, and intensity and size of the fire, which are driven by weather and climate conditions but also human influence. Prescribed fires conducted under conditions when fuel consumption is limited will minimise smoke emissions. Wildfires, in contrast, consume very large amounts of fuel with few or no options for reducing smoke impacts on populated areas (Yokelson *et al.* 2013).

In order to assess public health risk, it is essential to take into account not only the biome (fuel type) but also the burning conditions (i.e. dominance of flaming or smouldering combustion), fuel load and intensity of fire. Owing to the high fuel load, smouldering combustion and smoke composition, peat fires are likely to pose the greatest health risk. Tropical forests generally emit fewer pollutants per kilogram of biomass burned than boreal and temperate forests; however, owing to the high fuel load, emissions can be significant. Few research studies have focussed on the impact of tropical fires on air quality and future research should focus on the effects of smoke exposure from wildfires in tropical regions. Savannas have the lowest emissions of toxic compounds based on the lower EFs and predominantly flaming combustion.

External environmental factors (e.g. temperature, wind speed, wind direction and topography) and fire intensity influence smoke dispersion as well as duration of smoke exposure. Plumes from high-intensity fires can impact communities hundreds to thousands of kilometres away from the fire as plumes can be injected into the free troposphere and travel over vast distances. During transport, physical, chemical and photochemical processes occur, rapidly changing the composition of emissions and hence exposures to populations downwind. Furthermore, limited atmospheric mixing under stagnant night-time inversions can trap pollutants in valleys for extended periods of time, causing significant population exposures. Great uncertainty persists in determining plume distribution and chemistry, but satellite data and remote-sensing tools for forecasting smoke conditions near populated areas are continually being developed and improved and will greatly assist in both predicting and assessing public health risk (Yao *et al.* 2013; Yao and Henderson 2014).

Climatic conditions have also been shown to affect the behaviour, size and intensity of wildfires, with an increase in the severity and frequency of wildfires predicted with climate change (Keywood *et al.* 2013).

The main components of wildfire smoke that have the potential to cause adverse health effects are PM, CO, O_3, NO_2, SO_2, NMOCs and Hg. Exposures to high concentrations of these pollutants have been demonstrated to cause eye, nose or throat irritation, exacerbation of asthma, reduced lung function, bronchitis, cardiovascular disease and increased risk of cancer and mortality. However, the evidence for such effects from exposure to wildfire smoke is variable. There is a clear association with respiratory effects and mortality, but evidence for the other effects is limited (Youssouf *et al.* 2014; Dennekamp *et al.* 2015; Liu *et al.* 2015). Wildfires are variable events that can result in large short-term peak exposures locally, and longer-term stable exposures in distant communities of very short (hours) to longer duration (weeks). Health impacts are influenced by many factors including smoke composition, concentration of constituents, duration of exposure and underlying health status of the exposed population. The following summarises the current evidence.

- The major pollutant of concern in fire plumes is PM, which consistently exceeds air-quality guidelines. Particles can also be transported over long distances and impact on communities hundreds to thousands of kilometres away from the fire. Aged smoke plume particles are likely to have a higher organic content (Keywood *et al.* 2015) and potentially pose a greater health risk.
- High O_3 concentrations can also pose a potential health risk, in particular during warmer months in urban areas with elevated background O_3 concentrations. However, the impacts of fire plumes on O_3 ground concentrations can be quite variable and both increased and decreased concentrations of O_3 have been observed in fire-impacted areas (Alvarado *et al.* 2010). O_3 formation in smoke plumes is quite complex and depends on solar radiation, NO_x and NMOC concentrations, ageing of the plume and deposition processes. Generally higher enhancements are observed in tropical regions and in aged smoke plumes.
- Whereas the impact of NMOCs and other trace gases (NO_x) on health is minimal, emissions of NMOC and NO_x influence atmospheric chemistry and can significantly contribute to the formation of O_3 and SOA, thereby posing an indirect health risk.

- Emissions of NO_x and SO_x are highly dependent on the N and S content of the fuel. Biomass with higher N and S content includes peatlands. Large fires in these biomes can result in elevated concentrations of NO_2 and SO_2 and potentially cause exceedances of air-quality guidelines.
- CO is a stable tracer of combustion with enhanced concentrations (in the range of parts per billion) observed in fire plumes. However, enhancements in CO concentrations generally do not cause major air-quality issues. CO, however, is a pollutant of major concern on the fire line and close to fires.
- Hg can be a very dangerous contaminant owing to increased risk of cardiovascular disease and neurological impacts. Currently, there are insufficient data to fully understand the health impacts from exposure to Hg in wildfire smoke.
- Information on the production of free radicals during combustion, their persistence in the atmosphere and their potential to cause adverse health effects is scarce. Although studies have shown that free radicals in wood smoke may cause cellular and DNA damage, they may not persist in the atmosphere long enough to reach downwind communities. They may, however, be of concern to people exposed on the fire line.

Over the last few years, extensive research has been conducted on the impact of wildfire smoke on air quality, climate and health, but there still remains considerable uncertainty in certain areas. Emission inventories of components such as NMOCs and Hg as well as the contribution of NMOCs to SOA and O_3 formation are uncertain. New measurement techniques for trace gases and particle species have greatly improved emission characterisation but the spatial and temporal variability in emission factors and changes in burning conditions still present a challenge to accurately determine emissions.

Health studies from exposure to wildfire smoke have primarily focussed on PM, with limited knowledge of health effects associated with other smoke constituents. This is further complicated by the substantial uncertainty in validating standard levels of smoke components such as NMOCs to inform public health decisions. It is also important to know whether short-term exposures (hours) to very high concentrations of air pollutants trigger the health effects or whether the health impact is due to extended exposures (days–weeks) to pollutants. The health effects of populations regularly exposed to forest fire smoke are also unknown; however, understanding these is essential to examine the effects from smoke components with longer latencies.

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