The global impact of uncontrolled burning of waste

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Abstract

Open burning of solid waste is a significant source of air pollution emissions and a common practice in many regions of the world. Black carbon (BC) is a particularly important combustion emission pollutant having a global warming potential (GWP) up to 5000 times greater than carbon dioxide (CO_2) (Bond et al., 2013). However there are major uncertainties about the extent and impacts of the BC emissions from uncontrolled burning of waste on the environment. Specific waste BC emission factors (EFs) were measured in the laboratory to develop regional EFs based on the waste composition characteristics of different areas of the world. The results demonstrate that BC emissions from open burning of waste have a significant climate impact, contributing almost 5% of global CO_2Eq emissions. Global BC CO_2Eq emissions from burning waste are approximately 4 times larger compared to methane (CH_4) CO_2Eq emissions arising from the decomposition of equivalent amounts of combustible biodegradable waste disposed at dumpsites. Action to reduce open burning of waste would have a significant and immediate benefit to improving air quality and reducing climate change.

KEYWORDS: Open burning of waste, global estimates of black carbon emissions, air pollution, municipal solid waste and climate change.

1 Introduction

Open burning of solid waste in backyards and at uncontrolled dumpsites is a significant source of black carbon (BC), a short-lived climate pollutant (SLCP) that contributes to climate change, and which also has direct detrimental impacts on human health. However, BC emissions from open burning of waste are not included in most emission inventories used to model and develop national/international climate change mitigation policies. For example, BC is not considered in the IPCC Fourth (Bogner et al., 2007) or Fifth (IPCC, 2014) Assessment Reports.

To quantify the emissions from open burning of waste, it is necessary to use emission factors (EFs), which are described as the mass of a pollutant emitted per unit mass of material burned (Lemieux et al., 2004, Zhang and Morawska, 2002). In this research, EFs refers to the mass of BC emitted per kilogram of waste burned. Published BC EFs for open burning of waste are based on very limited field measurements and therefore have high levels of uncertainty. Large uncertainties in both the available EFs and the activity levels estimating the amounts of waste burned are major barriers to recognising the impact of this potentially significant emission source.

Wiedinmyer et al. (2014) highlighted the need to improve the available data on open burning of waste and that attention should be given to developing EFs and providing better estimates of burning waste. Waste

composition and the associated impact on the combustion characteristics have a significant effect on the nature and properties of the particulate and BC emissions from burning waste. Measuring EF values for specific waste types and combining this information to generate overall EFs for mixed waste with different compositions may offer a practical tool to estimate EFs for different local, regional or national waste sources.

One of the most effective ways to develop EFs for open burning sources is through laboratory simulations. However, EFs are significantly affected by the combustion conditions (Lemieux et al., 2004, Tissari et al., 2008, Zhang and Morawska, 2002). Therefore, we have developed an experimental protocol where small samples of waste were combusted in as representative and controlled a manner as possible and variables such as the mass of the sample, the mass of the emission captured, airflow rates, and the concentrations of emitted pollutants were carefully and critically measured or controlled. The EF is combined with the activity level, usually in terms of waste burned per unit time in a specific area, to calculate the overall emissions from open burning of waste.

The aim of the research presented in this paper was to develop regional EFs for BC from open burning of waste. The main objectives were to:

a) Develop a laboratory based, combustion methodology to measure BC EFs from burning waste;
b) Measure BC EFs for single waste types including: green waste, different types of plastics, textiles and paper and card;
c) Measure BC EFs for mixtures of these waste types for a representative developing country (Mexico);
d) Calculate regional EFs for BC from open burning waste based on the waste composition in different areas of the world; and
e) Place the BC emissions from open burning into context of other polluting emissions from the waste sector, such as methane (CH_4).

2 Material and methods

2.1 Sample collection

Black carbon characterization involves collecting particulate matter (PM) on filters either (1) by measuring the attenuation of light transmitted through the filter, or (2) by measuring the carbon content on the filter using thermo-optical analysis (TOA) (Watson et al., 2005). A laboratory waste emission sampler was designed and constructed as shown in Figure 1. The structure of the equipment consisted of a conic stainless-steel hood that was fixed to an aluminium stack equipped with a filter holder. A flowmeter was connected between the filter holder and a vacuum pump by a flexible hose. A metallic tripod was placed under the conical hood to hold porcelain crucibles containing waste samples during the experimentation process.

Small samples (0.1-5g) of single waste fractions (paper & cardboard, garden waste, textiles, LDPE, HDPE, PET, and polystyrene) were combusted in a representative manner in a laboratory fume cupboard using the waste emission sampler (Figure 1). The process involved placing the waste samples in a clean porcelain crucible on a tripod, a vacuum pump was attached to the sampling device and the airflow rate was recorded. The sample was ignited with a Bunsen burner, for a period of 10 seconds contact with the flame. The

combustion time of the experiment was recorded with a chronometer. The emissions from the combustion process were drawn through a 47 mm diameter Whatman quartz filter placed in the filter holder. The weight of the filters (pre-fired at 550ºC for 2.5 hours) was accurately measured with a four-digit balance before and after each experiment. The total emission mass was determined by the difference of weight in the filters. The crucible and sample were weighed at the end of the process to record the sample mass that was lost during combustion.

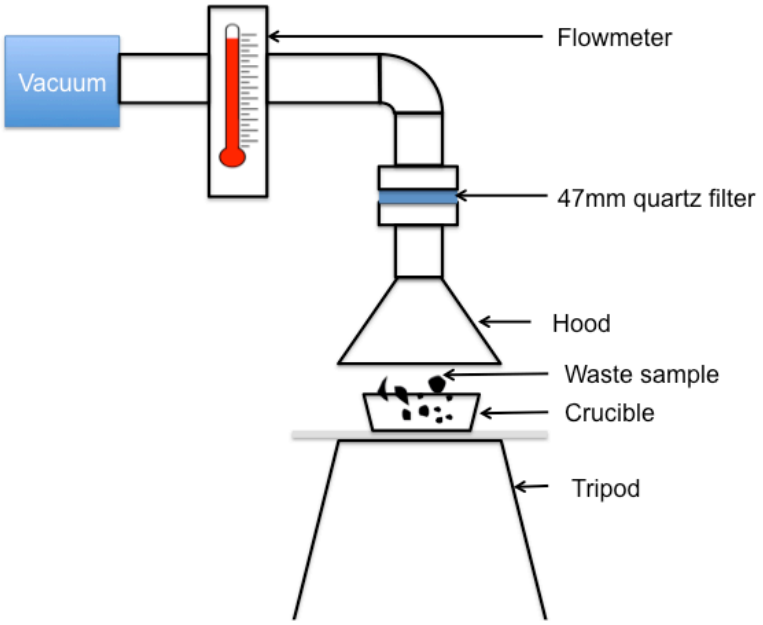

Figure 1 Waste combustion emission sampler

The weight of the samples was pre-determined by the initial visual inspection of the filters, as experience with measuring the light transmission properties of the residue showed that the optical transmission meter was saturated if the filter was too dark. Therefore, 3 to 5 pre-test burns were performed with different sample weights, and if the filter was too darkened, the sample weight was reduced. After determining the appropriate sample weight, 10 replicate burns with approximately equivalent sample weights were completed. The smallest sample weight used was 0.1 g for polystyrene, which produced the most dense emissions of all the different waste types tested. On the other hand, if there was little or no evidence of darkening on the filters in the initial pre-test experiments, the sample weight was increased to enhance the collection of particles on the filters. Thus, the largest sample size used was 5 g for PET and HDPE. The sample size used for LDPE was intermediate (3 g) due the lower density of this material compared to the other plastic materials examined. Green waste was dried in a furnace at 105ºC for 3 hours before the experimentation process, following the methodology of Shreve et al. (2006).

The variability of the combustion process prompted an analysis of the minimum replication frequency required for the experiments. Two waste types were selected including textiles and cardboard, to evaluate the variability in combustion emissions. The total mass of emissions captured on the filters for 20 samples of each

waste type (g of emissions per kg of sample) was determined. The standard deviation of increasing replications of the experiment was calculated by randomly selecting sample results from this data pool for each material. This exercise was repeated five times, and the overall mean stand deviation was calculated. The results of the statistical analysis showed that the mean stand deviation reached a stable value above ten replicates, as shown in Figure 2. Therefore, a minimum of 10 replicate test burns were completed for each waste sample.

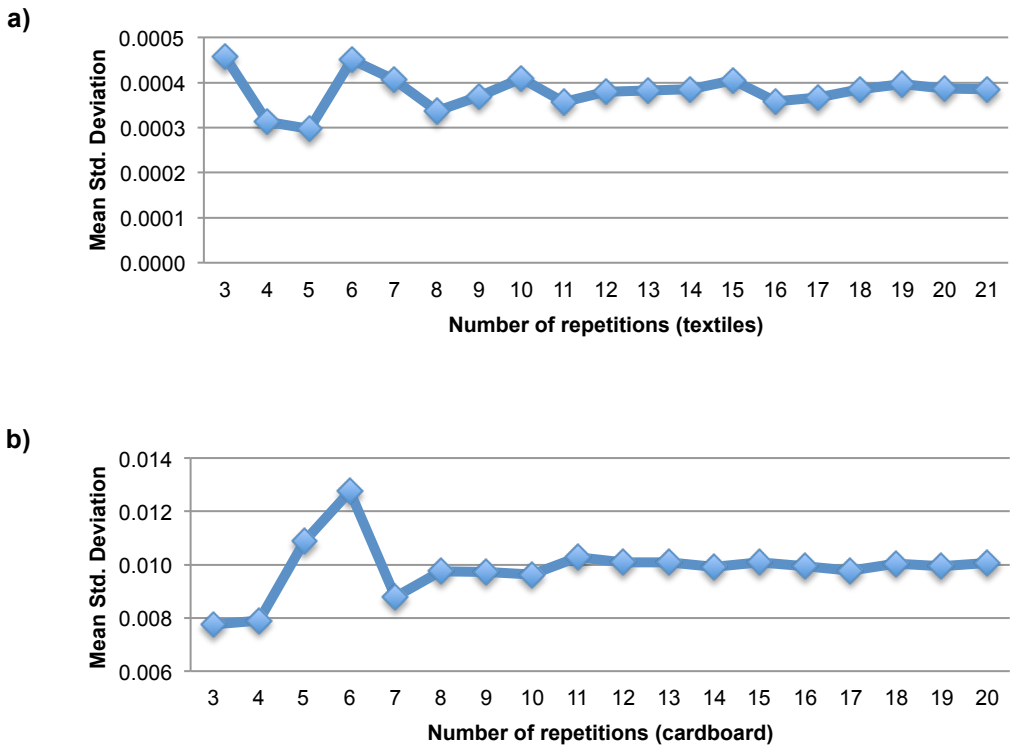

Figure 2 Mean standard deviation per number of repetitions for combustion emissions captured for (a) textiles and (b) cardboard

2.2 Determination of the carbon content of the samples

2.2.1 Determination of BC by optical analysis

Black carbon captured on the filters was characterized by measuring the attenuation of light transmitted through the filter using a SootScan™ Model OT21 Transmissometer; this is a non-destructive method and can perform rapid BC determinations. This instrument measures the transmission intensity of light passing through a particle-loaded filter relative to the intensity of a blank filter to determine the attenuation (ATN) value at two wavelengths (880 and 370nm). Infrared (IR) attenuation at 880 nm is considered to correspond to BC (Magee Scientific, 2015), therefore the results obtained for this wavelength were used to determine the attenuation of the filters. Sample sizes for combustion were adjusted to avoid saturation of the filter, however, where filters were saturated and a direct ATN value could not be taken, in these relatively small number of cases an ATN of 500 was assigned, which is the upper limit of detection of the OT-21 device.

The operation manual of the OT-21, states that 1 ATN unit is equivalent to 0.06 μg cm^{-2} BC on filters. According to Hansen (2005), ATN is given by the following equation:

$$ATN = 100 * \ln(I/Io) \quad \textbf{Equation 1}$$

where *I* and *Io* are the light intensities transmitted through the loaded and blank filters respectively (note that a scaling factor of 100 is used for numerical convenience). Optical methods assume that ATN is proportional to the BC on the filter paper (Ahmed et al., 2009, Hansen et al., 1984, Liousse et al., 1993, Petzold et al., 1997). BC density (in μg cm^{-2}) is calculated by dividing the ATN by the mass extinction coefficient (σATN), in $m^2\ g^{-1}$, as in the following equation (Hansen, 2005):

$$BC = ATN/\sigma ATN \quad \textbf{Equation 2}$$

This coefficient also referred to as "sigma", is not a constant, as it depends on the wavelength of the light used by the measuring instrument. The Magee Scientific Aethalometer user manual suggests a sigma value of 16.6 $m^2\ g^{-1}$ for BC measurements made using the IR wavelength channel (λ) = 880 nm (Hansen, 2005). This manual also suggests a sigma value of 12.6 $m^2\ g^{-1}$ when calibrated against elemental carbon (EC) measured using a thermal/optical reflectance (TOR) method. Hansen also states that if EC measurements were determined using the thermal/optical transmittance (TOT) method, such as in this project, a different sigma would be required, but did not provide a suggested value.

Several studies have stated that the rate of change in light transmission is not linear for filter samples, especially when ATN values are high due to a saturation effect (Davy et al., 2017, Drinovec et al., 2015, Virkkula et al., 2007, Weingartner et al., 2003). When saturation occurs, assuming linear proportionality between ATN and BC would lead to an underestimation of EC. Therefore, measurements must be compensated for this loading effect, to be able to accurately represent BC concentrations (Drinovec et al., 2015). Lack et al. (2014) summarise the three main factors that cause a non-linear response as follows:

- **Multiple light scattering within filters**: incident light may be scattered by unloaded filter substrate (membrane or fibres) and increase the sample path. This effect varies between different types of filter substrates.
- **Filter loading**: as more particles are collected on the filter the optical sampling path is decreased leading to bias in the final calculation. This effect will vary depending on the amount of material deposited on the filter.
- **Particle scattering**: as the filter becomes loaded there is an increase in the number of light scattering particles, which leads to incident light being scattered in any direction. This phenomenon could cause higher filter reflectance and more opportunities for light to be absorbed by other particles. This variation is dependent on the aerosol type (shape, size, composition, mass).

Therefore, BC measurements by ATN are adjusted to account for this loading effect using a quadratic equation (; Drinovec et al. (2015):

$$\mathbf{B}\ [\mu\mathbf{g}\ cm^{-2}] = \frac{k}{\sigma_{ATN}} * ATN^{2} + \frac{1}{\sigma} * ATN + 0 \qquad \textbf{Equation 3}$$

2.2.2 Determination of elemental carbon by thermal optical analysis

Elemental carbon (EC) and BC are operationally defined by the measurement method applied, although these two terms are often used interchangeably (Petzold et al., 2013, Watson et al., 2005). The thermal method strictly measures EC, but here we report the results as BC for consistency. Combustion-generated carbonaceous aerosol is measured either optically or chemically. The optical method uses light absorption, as in the case of the OT-21, and the resulting measurements of attenuation can then be converted to BC. The chemical (thermal) method determines the carbonaceous aerosol fraction of particulate matter samples, classifying it into organic carbon (OC) and elemental carbon (EC) (Cavalli et al., 2010, Karanasiou et al., 2015).

The European Committee for Standardization (CEN) has considered the thermal-optical analysis as the reference methodology to quantify EC and OC on filters. The most common protocols for this process include NIOSH-like, IMPROVE_ A and EUSAAR_2 either with transmittance or reflectance correction (Karanasiou et al., 2015). Karanasiou et al. (2015) reviewed the literature of these protocols and concluded that it is not possible to identify which one of these thermal optical methods is the “best”. They also conclude that all of these protocols are comparable for TC concentrations. However, an important discrepancy was found concerning OC and EC concentrations (Karanasiou et al., 2015). For this project, a Sunset Laboratory Aerosol Analyser was used to determine the elemental and organic carbon (EC/OC) content of the emission samples by the National Physical Laboratory (NPL) in the UK.

2.3 Estimating mixed waste EFs from the combustion properties of single waste streams

Emission factors were derived for mixed wastes by combining the EFs measured for individual waste types, based on the waste composition representative of three case study areas in Mexico. Derived EFs for mixed waste were compared to measured EFs for these different mixed waste streams. The derived EFs were on average, approximately 5 times larger compared to those measured in the laboratory, as shown in Table 1. Therefore we applied a nominal adjustment of 0.2 to calculate mixed waste EFs from the single waste stream data.

Table 1 Measured vs constructed EFs (g kg^{-1})

Mixed Waste types	Measured EFs (g/kg)	Derived EFs (g/kg)	EF Derived / EF Measured
Mexico	2.40	12.65	5.28
Huejutla, Hidalgo, Mexico	1.80	7.50	4.15
Juchitán, Oaxaca, Mexico	0.82	4.82	5.86

The results suggested that the combustion properties of individual materials varied greatly and produced much larger emissions compared to typical mixtures of waste in practice. Some of the factors that could influence waste combustion may be related to increased oxygen available in homogenous waste compared to

single waste types since poorly oxygenated, smouldering waste produces the highest emissions (Solorzano-Ochoa et al., 2012). This may be explained because homogenous waste samples appeared more dense and compacted compared to mixed waste samples, resulting in less oxygenated combustion conditions and thus, higher EFs. Further investigation is required to understand the interaction between different materials during the combustion process of single and mixed waste types.

A comparison between mixed waste BC EFs for the field case areas and published BC EFs is presented in Table 2. Although the experimental approaches were very different, for example, Christian et al. (2010) measured emissions in the field captured at three different landfill sites with different waste compositions, the results from field and laboratory were generally consistent. Thus, the average BC EF of the available published data was 1.98 g kg^{-1} compared to a mean value of approximately 1.7 g kg^{-1} measured in the laboratory.

Table 2 Measured EFs compared to published values (g kg^{-1})

Emission factors (g kg^{-1} waste)				
Number of test				
	Burn 1	**Burn 2**	**Burn 3**	**Average**
Christian et al. (2010)	0.38	0.92	0.63	0.65
Stockwell et al. (2016)	0.56 (wet conditions)	6.04 (dry conditions)	-	3.30
Jayarathne et al. (2017)	-	-	-	2.59*
Type of waste (n=repetitions)				
This research	**Mexico (n=11)**	**Huejutla (n=13)**	**Juchitán (n=16)**	**Average**
	2.40	1.81	0.82	1.67
*Three samples were measured but only the average EF was reported.				

2.4 Regional emission factors

Regional EFs were constructed using the single waste EFs obtained from the laboratory waste combustion experiments. These values were combined with information on waste composition in the different regions of the world published by the World Bank (Hoornweg and Bhada-Tata, 2012). The regions presented were: Africa Region (AFR), East Asia & Pacific (EAP), Eastern & Central Asia (ECA), South Asia (SAR), Middle East & North Africa (MENA), Latin America & the Caribbean (LCR) and countries belonging to the Organisation for Economic Co-operation and Development (OECD). The waste component categories used included: organic, paper & cardboard, plastics, glass, metals and others. The fraction of waste described as 'others' contained textiles, leather, rubber, dirt, multi-material packaging (such as tetra pack), e-waste and house appliances. This waste stream was considered to be mostly combustible waste. The World Bank study did not provide the composition of organic waste and therefore it was necessary to find additional information to determine the proportion of garden waste and food waste within this waste fraction. Alternative data to quantify the size of the textile waste fraction and the composition of the plastic fraction was also gathered. These data sources are presented in the supplementary material at the end of this document.

Regional BC EFs were calculated using Equation 4:

$$\textbf{Regional BC EFs} = [(GW_f * EF_{GW}) + (T_f * EF_T) + (PC_f * EF_{PC}) + (LDPE_f * EF_{LDPE}) + \quad \textbf{Equation 4}$$

$$(HDPE_f * EF_{HDPE}) + (PS_f * EF_{PS}) + (PET_f * EF_{PET})] * (0.2)$$

where the specific waste types included green waste (GW), textiles, (T), paper and card (PC), low and high density polyethylene (LDPE and HDPE, respectively), polystyrene (PS), and polyethylene terephthalate (PET); subscript 'r' represents the waste fraction and EF represents the emission factor for each waste type. Regional BC EFs were calculated by adjusting the measured EF values obtained for individual waste stream EFs by a factor of 0.2, as determined in section 2.3.

2.5 Global open burning of waste emissions

Global BC emissions from open burning of waste were calculated by combining the regional derived EFs with information on the activity levels for open burning of waste published by Wiedinmyer et al. (2014). The relative scale and impact of BC CO_2Eq emissions from open burning were compared to CH_4 CO_2Eq emissions arising from the decomposition of equivalent amounts and types of waste resulting from landfill disposal.

3 Results and discussion

3.1 Descriptive statistics of waste combustion and ATN

Table 3 shows the statistical examination for waste combustion and ATN values for single waste types. The fraction burned was highest for paper and cardboard, with 65% of the sample being combusted during the experiment. Organic waste and textiles also showed high levels of combustibility (57% and 51%, respectively). The materials with the lowest fraction burned corresponded to polystyrene and PET, with 21% and 10%, respectively. The rate of emissions captured per sample burned was highest for polystyrene (3%), followed by paper and cardboard (2%). Conversely, LDPE and HDPE presented the lowest rate of emissions capture (0.55% and 0.34% respectively). The highest values for ATN corresponded to PET (383), textiles (363) and polystyrene (328). LDPE and paper and cardboard presented the lowest ATN values for single waste types (46 and 62, respectively).

Table 3 Combustion patterns for single waste samples

	Fraction Burned (%)	Emissions Captured/Sample Burned (%)	Total Combustible Mass (%)	Remaining Volatile Fraction (%)	ATN
Paper and cardboard n=23 ATN>500, 0					
Mean	64.9	1.6	90.5	24.4	62
Std. Deviation	11.8	0.7	90.6	34.1	77
Garden n=15 ATN>500, 2					
Mean	57.1	0.9	93.7	41.6	264
Std. Deviation	18.5	0.6	93.3	73.6	122
Textiles n=14 ATN>500, 5					
Mean	51.1	1.1	99.7	48.8	363
Std. Deviation	9.2	0.4	100.0	49.2	114
LDPE n=17 ATN>500, 0					
Mean	23.8	0.3	84.3	59.8	46
Std. Deviation	6.1	0.1	84.4	60.5	67
HDPE n=20 ATN>500, 4					
Mean	40.2	0.6	99.9	58.9	147
Std. Deviation	28.2	0.5	99.9	90.2	193

		PET n=17 ATN>500, 9			
Mean	9.9	1.0	99.6	92.8	383
Std. Deviation	9.3	0.3	99.6	103.1	148
		Polystyrene n=14 ATN>500, 4			
Mean	21.3	3.0	100.0	79.5	328
Std. Deviation	7.7	2.2	100.0	88.3	162
ATN>500 n = number of filters marked as "Too Dark" by OT21					

3.2 Black carbon and ATN relationship: calculation of emission factors

The ratio of transmittance (ln I_0/I = ATN/100) and BC was non-linear and was represented by a quadratic equation with a high degree of statistical confidence (Figure 3).

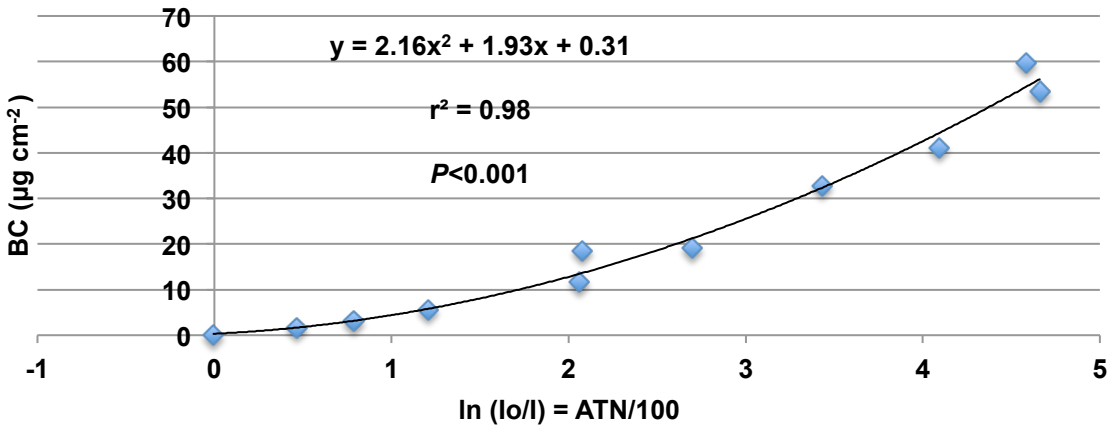

Figure 3 Relationship between ln (Io/I) and BC (µg cm^{-2}) determined by thermal optical transmission analysis

The consistent relationship between ATN and BC, measured by the thermal optical analysis method, is also demonstrated by the independent measurement of two samples with similar ATN, which gave very similar BC results.

3.3 Emission factors.

Emission factors were normalised (see Table 4) to be able compare values between different waste types (g of emissions per kg of waste burned). These calculations were carried out as follows:

1. The amount of BC per unit area (µg/cm^2) was calculated using ATN/100 measurements in the quadratic equation shown in Figure 3.
2. The value for BC per unit area was multiplied by the exposed area of the filter (9.08 cm^2) to obtain the total amount of BC emitted per waste sample.
3. A correction was applied to account for the volatile fraction of the sample that did not burn during the combustion process (Table 3). To do this, the BC per sample burned was calculated, and multiplied by the total volatile solid fraction of the sample.
4. Finally, the emissions per mass of waste burned were obtained in µg g^{-1}. This value was divided by 1000 to obtain an emission factor (EF) in g of BC per kg of waste burned.

Table 4 Mean BC EF for different waste types

Type of waste	EC (μg cm^{-2})	Total EC (μg)	EC/sample burned (μg g^{-1})	EC for total volatile solids (μg g^{-1})	EC EF (g kg^{-1})
Paper and cardboard	3.6	32.6	15.4	43.3	0.005 (±0.01)
Organic waste	23.5	213.0	276.2	344.7	0.245 (±0.21)
Textiles	38.4	348.7	4,659.5	663.9	42.032 (±27.75)
LDPE	2.6	23.4	31.0	71.4	0.013 (±0.02)
HDPE	15.4	139.5	66.7	192.0	0.035 (±0.04)
PET	43.7	397.2	1,784.0	6,777.1	0.715 (±0.80)
Polystyrene	35.1	318.6	10,569.1	1,568.0	92.462 (±81.33)

The highest EFs were obtained for polystyrene (92.5 g kg^{-1}) and textiles (42 g kg^{-1}). For these materials, even small amounts burnt in uncontrolled fires may result in very substantial emissions of BC. The lowest EF corresponded to paper and cardboard (0.005), followed by LDPE (0.013).

3.4 Regional EFs

The region emitting the largest amounts of BC from open burning of waste was East Asia and the Pacific (EAP), as shown in Table 5. In EAP, more than 441 x 10^3 t y^{-1} of BC are emitted from this source. Latin America and the Caribbean (LCR) were the next largest emitters of BC from burning waste at approximately 50% of the EAP value, equivalent to 210 x 10^3 t y^{-1}. As may be expected, the lowest BC emissions were in high-income OECD countries, despite them generating almost half of the world's waste; they also have the highest collection efficiency at 98 % (Hoornweg and Bhada-Tata, 2012). Moreover, open burning of waste in landfills has been eradicated in OECD countries. Nevertheless, over 10 x 10^6 t y^{-1} of waste are burned as a disposal method by households in OECD countries, releasing over 12.8 x 10^3 t y^{-1} of BC. In total, 1.1 x 10^6 t y^{-1} of BC are released from open burning of waste globally, equivalent to over 2,500 x 10^6 t y^{-1} of CO_2Eq. According to the World Bank (World Bank, 2018), the total emission of CO_2Eq in the world in 2012 was more than 53,500 x 10^6 t y^{-1}, therefore, open burning of waste represents approximately 4.7% of total global CO_2Eq emissions.

Table 5 BC, CH_4 and CO_2Eq emissions for different regions of the world (t y^{-1} unless otherwise stated)

					[(a)] EF CH_4 ($MTCO_2Eq$ Wet t^{-1})**				
					1.59	0.63	2.3		
	[(b)]Waste burned	BC Emission Factors (kg t^{-1})	Emissions		[(d)]Food waste***	Garden waste	Paper & cardboard	[(e)]CH_4 CO_2Eq emissions****	BC CO_2Eq / CH_4 CO_2Eq
			BC	[(c)]BC (CO_2Eq)					
Africa Region (AFR)	107,300,000	1.15	123,000	270,900,000	23,100,000	15,100,000	9,700,000	68,300,000	4
East Asia & Pacific (EAP)	396,600,000	1.11	442,000	971,400,000	99,100,000	47,600,000	39,700,000	278,500,000	3.5
Eastern & Central Asia (ECA)	99,200,000	1.21	120,000	264,600,000	9,700,000	27,200,000	13,900,000	64,400,000	4.1
South Asia (SAR)	153,800,000	1.16	179,000	393,900,000	31,600,000	13,600,000	6,200,000	72,900,000	5.4
Middle East & North Africa (MENA)	61,500,000	0.92	57,000	125,000,000	16,900,000	3,700,000	8,600,000	49,000,000	2.6
Latin America & the Caribbean (LAC)	143,700,000	1.47	211,000	463,400,000	28,000,000	21,500,000	23,000,000	110,900,000	4.2

Organisation for Economic Co-operation and Development (OECD)	10,100,000	1.27	13,000	28,300,000	600,000	1,600,000	3,200,000	9,400,000	3
WORLD	972,200,000	-	0	2,517,500,000	209,000,000	130,300,000	104,200,000	653,300,000	3.8 (average)

a) Emission factors for CH_4 in metric tonnes of CO_2 equivalent per wet ton of waste are taken from the Waste Reduction Model (WARM) (US EPA, 2015).
b) Source: Wiedinmyer et al. (2014).
c) Average global warming potential (GWP) for BC in a 20 year horizon is 2200 (Bond et al., 2013, Fuglestvedt et al., 2010).
d) It was considered that 50% of the food waste is burned as feeding waste to animals and using food waste for compost is a common practice in developing countries.
e) Methane emissions from equivalent disposal of waste fractions in landfill.

Methane CO_2Eq emissions for waste that is currently burned were also calculated for the different regions. The results showed that BC CO_2Eq from burning were four times larger than CH_4 CO_2Eq emissions from disposing of equivalent amounts of biodegradable waste in landfill. The largest impact was found in the South Asia region (SAR), where burning waste produces 5 times more CO_2Eq emissions than would occur if the waste were disposed by landfilling. The smallest difference was found for the Middle East and North Africa region, where CO_2Eq emissions from burning were three times larger than for landfill disposal.

4 Conclusions

We have developed BC EFs relevant to the open, uncontrolled burning of waste based on a small-scale, laboratory experimental combustion system. The EFs measured by this approach were consistent with the available limited field measurements of open burning of waste. Black carbon EFs from individual and mixed waste streams were produced and were applied to estimate the emissions from different regions of the world based on the characteristic waste composition from these areas. The results demonstrated that BC emissions from open burning of waste have a significant climate impact, contributing almost 5% to global CO_2Eq emissions. Global BC CO_2Eq emissions from burning waste may be approximately 4 times larger compared to CH_4 CO_2Eq emissions arising from the decomposition of equivalent amounts of combustible biodegradable waste disposed at dumpsites. Action to reduce open burning of waste would therefore have a significant and immediate impact on improving air quality and respiratory health, and, reducing climate change. The results presented here suggest the uncontrolled burning of waste is a much larger contributor to climate change compared to the CH_4 emissions from landfill disposal of biodegradable waste, which has received much attention. By contrast, open burning is not considered by the IPCC due to uncertainty about the contribution of this source, therefore, current climate change inventories significantly underestimate the global emissions from, and impacts of, the waste management sector.

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6 Supplementary Material

Sources for detailed information on waste composition

Regions	Garden Waste fraction in 'organics'		Textile fraction in 'others'		Specific plastic fractions	
	Area	Sources	Area	Sources	Area	Sources
AFR	Ghana and Gaborone, Botswana	(Miezah et al., 2015, Nagabooshnam, 2011)	Gaborone, Botswana	(Nagabooshnam, 2011)	Gaborone, Botswana	(Nagabooshnam, 2011)
EAP	Penang, Malaysia	(UNDP, 2008)	Penang,	(UNDP, 2008)	Bangkok, Thailand	(Areeprasert et al.,

			Malaysia			2017)
ECA	St. Petersburg, Russia	(Selezneva, 2016)	St. Petersburg, Russia	(Selezneva, 2016)	St. Petersburg, Russia	(Selezneva, 2016)
SAR	Lahore, Pakistan	(Kamran et al., 2015)	Lahore, Pakistan	(Kamran et al., 2015)	India	(Singh et al., 2017)
MENA	Israel and Jordan	(Ayalon et al., 1998), (Jordan Green Building Council, 2016) and (Abu-Salah et al., 2011)	Jordan	(Abu-Salah et al., 2011) and (Jordan Green Building Council, 2016)	Jordan	(Abu-Salah et al., 2011) and (Jordan Green Building Council, 2016)
LCR	Mexico	(SEMARNAT, 2013) and (INECC, 2012)	Mexico	(SEMARNAT, 2013) and (INECC, 2012)	Mexico	(SEMARNAT, 2013) and (INECC, 2012)
OECD	USA	(Subramanian, 2000) and (US EPA, 2013)	USA	(Subramanian, 2000) and (US EPA, 2013)	USA and EEUU	(Subramanian, 2000), (US EPA, 2013) and (European Commission: DG Environment, 2011)