**Experimental measurements of black carbon emission factors to estimate the global impact of
uncontrolled burning of waste**

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13 **Abstract**

14 Open burning is a widely practiced method of solid waste disposal in many regions of the world and represents
15 a significant source of air pollution. Black carbon (BC) is a particularly serious air pollutant emitted from the
16 uncontrolled burning of waste in open fires because it has a global warming potential (GWP) up to 5,000 times
17 greater than carbon dioxide (CO₂) and is also linked to detrimental health impacts. However, few quantitative
18 measurements of BC from open burning have been completed to establish the extent and impacts of this emission
19 source on the environment. Emission factors (EFs) for BC from burning mixed solid waste samples were measured
20 in the laboratory based on waste compositions in a representative developing country (Mexico). Black carbon EFs
21 were also derived for individual waste types, including: green waste, different types of plastics, textiles and paper
22 and cardboard. Individual waste BC EFs were combined using waste composition data from different areas of the
23 world to estimate regional and global BC emissions from this source. The results demonstrated that BC emissions
24 from open burning of waste have a significant climate impact, equivalent to 2 - 10 % of global CO₂Eq emissions.
25 Global BC CO₂Eq emissions from burning waste are 2 to 8 times larger compared to methane (CH₄) CO₂Eq
26 emissions arising from the decomposition of equivalent amounts of combustible biodegradable waste disposed at
27 dumpsites. Action to reduce open burning of waste would have a significant and immediate benefit to improving
28 air quality and reducing the potential impact on climate change.

29 **KEYWORDS:** Open burning, waste, black carbon, air pollution, emission factors, climate change.

30 **Highlights**

- 31 • BC EFs were measured for single and mixed waste streams.
- 32 • BC EFs were in the range of 0.02-53.1 g kg⁻¹, depending strongly on waste type.
- 33 • BC emissions from open burning of waste were calculated for different regions.
- 34 • BC emissions from burning waste contributes over 2-10% of global CO₂Eq emissions.
- 35 • BC CO₂Eq from burning is 4 times larger compared to CH₄ from waste decomposition.

36

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41 costs to support the fieldwork.

42

43 1 Introduction

44 Solid waste disposal is a worldwide issue and represents a severe risk to public health and the environment.
45 Globally, two billion people lack access to municipal solid waste (MSW) collection services and dispose of
46 household waste typically by open burning, burial or dumping on open ground or into watercourses (UNEP and
47 ISWA, 2015). Open burning of solid waste in backyards and at uncontrolled dumpsites is a significant emission
48 source of black carbon (BC), a short-lived climate pollutant (SLCP) that contributes to climate change mainly by
49 absorbing solar radiation and hence having a warming effect (UNEP, 2011). BC also has direct detrimental impacts
50 on human health (Janssen et al., 2012). However, BC emissions from open burning of waste are not included in
51 most emission inventories used to model and develop local/national/international climate change mitigation
52 policies. For example, BC is not considered in the IPCC Fourth (Bogner et al., 2007) or Fifth (IPCC, 2014)
53 Assessment Reports.

54 Quantifying emissions from open burning of waste requires accurate measurement of the emission factors
55 (EFs) of different environmental pollutants, which represent the mass of a specific pollutant emitted per unit mass
56 of material burned (Zhang and Morawska, 2002; Lemieux et al., 2004). Published BC EFs (Christian et al., 2010;
57 Stockwell et al., 2016; Jayarathne et al., 2017) for open burning of waste are based on very limited field
58 measurements and therefore a high degree of uncertainty is attached to them. A combination of large uncertainties
59 associated both with the available EFs for BC for open burning and the activity levels regarding the actual amounts
60 of waste burned by households in backyards and municipalities at dumpsites are major barriers to interpreting the
61 impact of this potentially significant emission source.

62 Field measurements of BC emissions from open burning present several technical and experimental challenges
63 and require the deployment of expensive monitoring equipment close to the combustion source to sample emission
64 concentrations within the fire plume (Yokelson et al., 2013; Aurell et al., 2017). It is also difficult to quantify the
65 overall mass balance necessary to derive a BC EF for open-burning of waste under field conditions for two main
66 reasons: firstly there may be uncertainty about the total amount of waste that is available for burning and, secondly,
67 the concentrations of pollutants in the plume are highly variable and strongly influenced by local meteorological
68 conditions. An alternative approach that has been widely applied to develop EFs for different burning materials, is
69 to use carefully controlled laboratory simulations (Ortiz-de-Zárate et al., 2000; Lemieux et al., 2004; Park et al.,
70 2013). EFs are significantly affected by the combustion conditions, but these can be better regulated and
71 monitored, and the emissions more effectively captured, in a laboratory compared to open burning tests in the field
72 (Zhang and Morawska, 2002; Lemieux et al., 2004; Tissari et al., 2008). Indeed, several authors (Yokelson et al.,
73 2008; McMeeking et al., 2009; Yokelson et al., 2013) demonstrate the consistent relationship between laboratory-
74 derived and field-based EFs from open burning of biomass. From a detailed comparison of laboratory data and
75 field measurements for prescribed fires, Yokelson et al. (2013) concluded that laboratory based experimentation
76 can significantly increase current knowledge of biomass combustion processes and emissions.

77 Black carbon characterization involves collecting particulate matter (PM) on filters and either: (1) measuring
78 the attenuation (ATN) of light transmitted through the filter and converting to a BC concentration, or (2) measuring
79 the elemental carbon (EC) content on the filter using thermal-optical analysis (TOA) (Watson et al., 2005). The
80 transmissometry technique measures the transmission intensity of light passing through a particle-loaded filter

relative to the intensity of a blank filter to determine the ATN value at two wavelengths (880 and 370nm). Infrared (IR) attenuation at 880 nm is considered to correspond to BC (Magee Scientific, 2015). The advantages of this technique is that it is rapid, straight-forward and not destructive, it can be performed with minimal operating and consumable cost and is therefore practical and suitable for measuring large numbers of experimental samples. A disadvantage of the ATN approach is that the transmission response is not linear at high PM densities on the filters and under these circumstances the relationship between BC and ATN requires calibration against a direct measure of BC concentration for the range of ATN values observed and the sample type under investigation. This can be done by TOA, which is the reference method recommended by the European standardization body (CEN) to quantify EC on discrete atmospheric PM collection filters (Karanasiou et al., 2015). In TOA, carbonaceous aerosol fractions are thermally desorbed from the filter medium under an inert helium atmosphere followed by an oxidizing atmosphere using controlled heating ramps and EC as well as organic carbon (OC) are determined by a flame ionization detector (FID) (Cavalli et al., 2010; Karanasiou et al., 2015). Whilst providing a direct, quantitative measurement, this approach requires a higher degree of specialism, equipment and expertise. Consequently, it is not routinely available and tends to be a cost intensive procedure that is only provided by a limited number of specialist geophysical laboratories. Thermal-optical analysis specifically measures the EC fraction of the collected PM and is considered as being operationally equivalent to BC for data interpretation purposes in atmospheric pollution research (Watson et al., 2005; Petzold et al., 2013; Yelverton et al., 2014).

Wiedinmyer et al. (2014) argued that reducing uncertainty and increasing availability of data on open burning of waste was an urgent priority, 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, has a significant effect on the nature and properties of the particulates, including BC emissions from burning waste. Measuring EF values for individual waste types and combining this information to generate overall EFs for waste streams of different compositions may offer a practical tool to estimate EFs for different local, regional or national waste sources without the need for multiple experiments on many different waste mixture combinations. This would facilitate greater accountability of the BC emissions from burning waste across local, regional and global levels.

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 emissions captured, airflow rates, and the concentrations of emitted pollutants were carefully and critically measured or controlled. The EF is combined with the activity level (which defines the amount of waste burned), usually in terms of waste burned per unit time in a specific geographic area, to calculate the overall emissions from open burning of waste.

The aim of the research discussed here is to measure BC EFs for representative waste streams which are characteristic of a developing country using Mexico as an example, and to use the experimental data to formulate proposed BC EFs from open burning of waste at a regional and global scale. The main objectives were to:

- a) develop a laboratory based, controlled methodology to capture BC emissions from waste combustion;
- b) measure captured BC emissions using a light ATN transmissometer technique and calibrated by TOA;
- c) determine BC EFs for mixtures of waste for a representative developing country (Mexico);

- d) determine BC EFs for individual waste types, including: green waste, different types of plastics, textiles and paper and cardboard;
- e) calculate regional EFs for BC from open burning of waste using single waste BC EFs based on the waste composition in different areas of the world;
- f) combine regional BC EFs with published activity levels on open burning of waste to estimate regional and global BC emissions from this source;
- g) establish the relative climate impact of BC emitted from open burning of waste compared to methane (CH₄) released from dumpsite disposal.

2 Material and methods

2.1 Selection of case study areas to derive mixed waste BC EFs

Mexico has one of the largest rates of residential and open dump burning of waste compared to other regions internationally and therefore, is potentially amongst the leading emitters of BC from MSW disposal globally (Wiedinmyer et al., 2014). Two case study areas in Mexico were selected for investigation as part of a programme of research to determine the extent and impact of open burning of waste in representative, contrasting urban and rural communities in developing countries, with different levels of waste service provision and waste composition analysis, and where open burning is widely practiced (Reyna-Bensusan et al., 2018; Reyna-Bensusan, 2019). This involved a programme of fieldwork, which amongst other data, gathered information on waste disposal, burning activity and composition analysis (Reyna-Bensusan, 2019). Figure 1 shows the location of the case study areas selected for the research, which were located in the Municipality of Huejutla, in the state of Hidalgo, and the Municipality of Juchitán, in the State of Oaxaca.

Figure 1 Case study areas selected for the research

2.2 Sample collection

A laboratory waste emission sampler was designed and constructed as shown in Figure 2. The structure of the equipment consisted of a conical stainless-steel hood fixed to an aluminium stack equipped with a filter holder, to capture and retain emitted PM. 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 support a porcelain crucible, which contained

145 the burning waste sample. The apparatus was located in a laboratory fume cupboard to conduct the combustion
146 experiments. The airflow applied by the vacuum pump was maintained at a constant rate equivalent to 40 l min⁻¹.

Figure 2 Waste combustion emission sampler

150 Small samples (0.1-5g) of mixed waste based on the waste composition in the case study areas (Reyna-
151 Bensusan et al., 2018; Reyna-Bensusan, 2019), and single waste fractions, were transferred to clean, dry
152 porcelain crucibles (15 cm Ø). The crucibles were prepared by heating to 600°C in a muffle furnace for at least 6
153 hours to remove residual matter and were stored in a laboratory desiccator before each experiment. The materials
154 used for the samples were common items disposed by open burning (including plastic bottles and plastic carrier
155 bags). The source of each material and the pre-combustion treatment received before the experimentation process
156 is described in Table 1. Mixed waste samples were prepared using information gathered in the field composition
157 analysis for the case study areas (Reyna-Bensusan et al., 2018; Reyna-Bensusan, 2019) and in official sources
158 for the national waste composition in Mexico (INECC, 2012; SEMARNAT, 2013a). Further details of these
159 procedures are given in Section 2.8.

Table 1 Sample materials and treatment

Waste type		Source	Treatment
Mixed waste samples	National	National waste composition was based on: (INECC, 2012; SEMARNAT, 2013a).	Wastes were sourced and treated as described for single waste samples in this table and combined to reflect the waste composition nationally and in the case study areas
	Huejutla	Waste composition analysis obtained from (Reyna-Bensusan, 2019).	
	Juchitán		
Single waste samples	Paper and cardboard	Mixed used paper and cardboard boxes.	Shredded to approximately 1cm ² particle size using a guillotine.
	Garden waste	Tree leaves and branches in a 60-40% mass proportion respectively (INECC, 2012).	Oven dried at 105°C for 3 hours (Shreve et al., 2006)
	Textiles	50% synthetic, 50% cotton based on the composition of textile waste from (INECC, 2012).	Shredded to approximately 1cm ² particle size using a scissors.
	Low density polyethylene (LDPE)	Plastic carrier bags	
	High density polyethylene (HDPE)	Plastic containers	
	Polyethylene terephthalate (PET)	Transparent water bottles	
	Polystyrene	Food trays	

161

162 The sample was ignited with a Bunsen burner by exposing the materials to the flame for a contact period of 10
163 seconds. The combustion time of the experiment was recorded with a chronometer. The materials were processed
164 following this consistent approach to ignite the waste and the subsequent combustion patterns were similar to the
165 behaviour observed for burning waste under field conditions, comprising both flaming and smouldering phases.
166 The emissions from the combustion process were drawn through a 47 mm diameter Whatman quartz filter. This
167 type of filter was chosen as it could be used in both light attenuation and TOA measurement methods. The filters
168 were pre-fired at 550°C for 2.5 hours in a muffle furnace and stored in a laboratory desiccator. Each filter was
169 accurately weighed with a four-digit analytical balance (Oertling VA 300) before and after each experiment. The
170 total mass of collected PM was determined by the difference in weight in the filters. The crucible and sample were
171 also weighed at the end of the process to record the sample mass that was lost during combustion.

172 2.3 Determination of sample weight

173 Visual inspection of the filters after combustion was used as a guide to adjust the sample weight to avoid
174 saturating the optical transmission (see Section 2.5 below for detail). Three to 5 pre-test burns were performed
175 with different sample weights for each waste type, and if the filter was significantly darkened, the sample weight
176 was reduced. After determining the appropriate sample weight, 10 replicate burns with equivalent sample weights
177 were completed. Polystyrene produced the most blackening of the emission collection filters compared to the other
178 waste types examined and the sample weight used for this material was 0.1 g. If there was little evidence of
179 darkening on the filters in the initial pre-test experiments, the sample weight was raised to increase the collection
180 of particles on the filters. Thus, the largest sample weight was 5 g for PET and HDPE and for LDPE the sample
181 weight was intermediate and equivalent to 3 g.

182 2.4 Determination of replication frequency

183 The variability of the combustion process prompted a replication analysis to determine the minimum number of
184 replicates required for the experiments. Two waste types were selected (textiles and cardboard) and 20 individual
185 combustion experiments were completed for each waste. The total mass of emissions captured on the filters for

each waste type (g of emissions per kg of sample) was determined. The standard deviation (sd) was calculated with increasing numbers of replicates (n=2 to 20) based on randomly selected sample results from the total data pool for each material. This exercise was repeated five times (for n<20) to provide an overall mean sd. The results showed that the mean sd was minimised above 10 replicates, which was selected as the minimum number of replicate combustion tests required for each waste type (see more details in the Supplementary Materials Section).

2.5 Determination of the BC content of the samples

Black carbon captured on the quartz filters was characterized by measuring the ATN of light transmitted through the filter using a SootScan™ Model OT21 Transmissometer. ATN is linear within the range of 0-125 ATN units, however, the transmissometer can measure ATN values up to ≈500 units, although filter saturation can produce a curvilinear response when ATN is >125 units (Magee Scientific, 2015). The sample weight (quantity of waste burned) used in the combustion experiments was therefore adjusted within the ATN detection range of the instrument. Nevertheless, for a small number of samples it was not possible to take a direct ATN measurement due to filter saturation with PM and, in these cases, the maximum ATN value of 500 units was assigned to the sample.

According to Hansen (2005), ATN is given by the following equation:

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

201

where I and I_o are the light intensities transmitted through the loaded and blank filters, respectively (a scaling factor of 100 is used for numerical convenience as suggested in the users manual by Hansen (2005)). Optical methods assume that ATN is proportional to the BC on the filter paper (Hansen et al., 1984; Lioussse et al., 1993; Petzold et al., 1997; Ahmed et al., 2009):

$$BC = ATN/\sigma_{ATN} \quad \text{Equation 2}$$

Black carbon measurements by ATN are adjusted to account for particulate loading effects on the filters using the quadratic equation (Virkkula et al., 2007; Greilinger et al., 2019):

$$BC \left[\mu g cm^{-2} \right] = \frac{k}{\sigma_{ATN}} * ATN^2 + \frac{1}{\sigma_{ATN}} * ATN \quad \text{Equation 3}$$

where, σ_{ATN} is the mass extinction coefficient, in $m^2 g^{-1}$, and k is the shadowing correction factor.

2.6 Determination of EC by thermal optical analysis

The TOA analysis of the PM filter samples was completed by the National Physical Laboratory (NPL) in the UK with the purpose of quantifying the calibration relationship between BC and ATN. Five filters covering the range of ATN measurements detected by the transmissometer were selected for TOA from the combustion tests using the mixed waste samples prepared for each case study area. For this analysis, a rectangular punch was taken from each filter and the EC content was determined using a Sunset Laboratory Aerosol Analyser, following the EUSAAR2 protocol with pyrolysis correction based on transmittance (Cavalli et al., 2010; Karanasiou et al., 2015).

216 Strictly speaking, the thermal-optical method measures EC, which may be considered as equivalent to BC
217 (Briggs and Long, 2016), therefore we report the EC results from the TOA as equivalent to BC following the advice
218 of Petzold et al. (2013) and to remain consistent with other atmospheric pollution research that refers to BC and
219 that follows a similar experimental approach (Bond et al., 2013; Long et al., 2013). The relationship between light
220 ATN In (Io/I) and BC is not linear, particularly at high particle densities, and was therefore determined empirically
221 by polynomial regression analysis using the statistical function of the Microsoft EXCEL computer programme.

222 **2.7 Calculating Emission Factors**

223 The emission factors were calculated by the following procedure:

- 224 1. The mass of BC per unit area on the filter ($\mu\text{g cm}^{-2}$) was calculated as described in Section 2.5.
- 225 2. The mass of BC per unit area was multiplied by the exposed area of the filter (9.08 cm^2) to obtain the
226 total mass of BC emitted per waste sample.
- 227 3. Observations made during the field investigation reported by Reyna-Bensusan et al. (2018) showed
228 that backyard burning takes place in the same area and dumpsites are continuously burning. Therefore,
229 all combustible material in waste will potentially contribute to the emission of BC. In the controlled
230 laboratory tests, it was apparent that burning ceased before all the combustible material was consumed,
231 after removal of the ignition source. Therefore, a correction was applied to account for the combustible
232 fraction of the sample that did not burn during the laboratory combustion test to adjust EFs to
233 correspond to the total combustible fraction in the waste. To do this, the mass of BC per sample burned
234 was calculated, and adjusted for the total combustible fraction of the sample. The total average
235 combustible fraction for each waste type was determined by ashing the samples in a muffle furnace at
236 600°C for 5 h following Sluiter et al. (2008) and ISO standard method 3451 (ISO, 2008).
- 237 4. Finally, a normalised mass of the emissions released per unit mass of waste burned was obtained in
238 $\mu\text{g g}^{-1}$. This value was divided by 1000 to obtain an emission factor (EF) in g of BC per kg of waste
239 burned.

240 **2.8 Estimating mixed waste EFs from the combustion properties of single waste streams**

241 The representative waste mixtures used for the combustion tests (see Section 2.2) were prepared by combining
242 the four main waste types in the appropriate proportions found in each region (Reyna-Bensusan et al., 2018;
243 Reyna-Bensusan, 2019) and reported nationally (INECC, 2012; SEMARNAT, 2013a), including organic matter,
244 textiles, paper and cardboard, and plastics (comprising the four major types: LDPE, HDPE, polystyrene and PET,
245 also in proportion). Minor residual waste materials (for example: cotton, leather, electrical waste, and rubber,
246 amongst others) each represented $\leq 1.0\%$ of the total waste mass and the mass of the main fractions examined in
247 the combustion tests were balanced to 100% and the mass fractions were distributed as shown in Table 2.

248

Table 2 Composition of mixed waste samples (% dry mass)

Waste Fraction	National (Mexico)[±]	Huejutla	Juchitán
Garden waste	25.5	43.5	51.8
Textiles	16.3	13.2	7.0
Paper and cardboard	30.0	17.3	12.7
Plastic 1: LDPE	10.9	10.2	10.6
Plastic 2: HDPE	5.6	4.8	5.2
Plastic 3: Polystyrene	6.2	1.9	1.8
Plastic 4: PET	5.6	9.0	10.9
TOTAL	100	100	100
*Source: (INECC, 2012; SEMARNAT, 2013b)			

249

250 Emission factors for mixed waste for both study areas and the national average for Mexico, were derived using
251 the specific BC EFs measured for individual waste types (paper and cardboard, organic, textiles and different
252 plastics), based on the compositional proportions listed in Table 2, and these are referred to as derived BC EFs.
253 Derived mixed waste EFs were compared with measured EF values obtained for the mixed waste samples to
254 establish a relationship between these values. Deriving EFs for mixed waste compositions from single waste
255 components has the potential to provide local, regional or national emissions estimates based on a simple
256 compositional analysis of the waste, avoiding the challenge and complexity of measuring the emissions from many
257 different and specific waste sources.

258 **2.9 Calculation of mixed waste EFs**

259 Mixed waste EFs for different regional areas were constructed using the single waste EFs obtained from the
260 laboratory waste combustion experiments. These values were combined with information on waste composition
261 in the different regions of the world published by the World Bank (Hoornweg and Bhada-Tata, 2012). The regions
262 presented were: Africa Region (AFR), East Asia & Pacific (EAP), Eastern & Central Asia (ECA), South Asia (SAR),
263 Middle East & North Africa (MENA), Latin America & the Caribbean (LCR) and countries belonging to the
264 Organisation for Economic Co-operation and Development (OECD) (see Figure 3). The waste component
265 categories used included: organic, paper and cardboard, plastics, glass, metals and others. The fraction of waste
266 described as ‘others’ contained textiles, leather, rubber, dirt, multi-material packaging (such as tetra pack), e-waste
267 and house appliances. This waste stream was considered to be mostly combustible waste. The World Bank study
268 did not provide the composition of organic waste and therefore it was necessary to find additional information to
269 determine the proportion of garden waste and food waste within this waste fraction. Alternative data to quantify
270 the size of the textile waste fraction and the composition of the plastic fraction was also gathered. Data sources to
271 determine these specific waste fractions are presented in the Supplementary Material.

Figure 3 Global Economic Regions (Hoorweg and Bhada-Tata, 2012)

Mixed waste BC EFs were derived from the combustion emissions measured for single waste types, based on the proportional contributions from each waste fraction according to the compositional analysis, using the following equation:

$$\begin{aligned} \text{Mixed waste BC EFs} = & [(GW_f * EF_{GW}) + (T_f * EF_T) + (PC_f * EF_{PC}) + (LDPE_f * EF_{LDPE}) \\ & + (HDPE_f * EF_{HDPE}) + (PS_f * EF_{PS}) + (PET_f * EF_{PET})] \end{aligned} \quad \text{Equation 4}$$

where the specific waste types included green waste (GW), textiles, (T), paper and cardboard (PC), low and high-density polyethylene (LDPE and HDPE, respectively), polystyrene (PS), and polyethylene terephthalate (PET); subscript 'f' represents the waste fraction and EF represents the emission factor for each waste type.

2.10 Global emissions from open burning of waste

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 provided by Wiedinmyer et al. (2014), using the following equation:

$$BC = W_B \times EF / 1000 \quad \text{Equation 5}$$

where BC is the mass of BC in t, W_B is the mass of waste burned in t and EF is the emission factor in g per kg of waste burned; a multiplication factor of 1000 is applied to convert kg emitted to t of BC.

The following equation was applied to determine the BC CO₂Eq:

$$BC\ CO_2Eq = BC \times GWP$$

Equation 6

where *GWP* is a factor representing the GWP of BC relative to CO₂. Arguably, it is inappropriate to represent the GWP value for BC as a single value because of its physical properties and it is not homogeneously mixed in the atmosphere. Therefore, we have applied the range of published values for the GWP of BC, equivalent to 1,200 and 5,100 (mean = 2,200) in a 20-year horizon (Fuglestvedt et al., 2010; Bond et al., 2013), and this also helps to capture the potential variability that may be associated with scaling up laboratory-based emission measurements for BC. The relative scale and impact of BC CO₂Eq emissions from open burning were compared to CH₄ CO₂Eq emissions arising from the decomposition of equivalent amounts and types of waste by landfill or dumpsite disposal. This analysis used landfill CH₄ CO₂Eq EFs for different major and representative waste material types (specifically, food waste, paper and cardboard and yard trimming fractions) provided by the Waste Reduction Model (WARM) (US EPA, 2015).

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 combustibility of a waste material was determined by the fraction of the total mass of waste that was destroyed by burning. The characteristics of the combustion patterns for mixed samples were strongly related to the composition of the waste for each case study area. The largest fraction of waste destroyed by burning was obtained for the average waste composition for Mexico as a whole. The emissions captured (determined by the mass difference of the filters before and after combustion) per sample burned (calculated by dividing the mass of emissions captured by the mass of the sample burned) and the ATN values were also larger for this mixed waste sample compared to the other representative regional samples for Mexico. This is explained because the national average waste composition contained a larger proportion of textiles and polystyrene, equivalent to 16 % and 7 % respectively, whereas the regional Huejutla and Juchitán mixed waste samples contained 13 % and 7 % of textiles, respectively, and 2 % of polystyrene. As single waste samples, textiles and polystyrene showed highest degrees of combustibility and ATN compared to the other waste types examined. Mixed samples based on the waste composition of Huejutla municipality produced more captured emissions relative to the amount of sample burned, and higher ATN values (367) compared to Juchitán (ATN = 300). This can be explained because the waste composition for Huejutla contained almost twice the amount of textiles compared to Juchitán (13 % and 7 %, respectively), although the waste from these communities contained generally a similar proportion of polystyrene (1.9 % and 1.8 %, respectively).

For single waste samples, the largest fraction burned was obtained for paper and cardboard, equivalent to 65 % of the initial sample mass. Garden waste and textiles also showed a high degree of combustibility (57 % and 51 %, respectively). The materials with the lowest fraction burned corresponded to polystyrene and PET, with 21 % and 10 %, respectively. Interestingly, however, the largest proportion of particulate emissions captured per sample burned was measured for polystyrene (3 %), followed by paper and cardboard (2 %). Conversely, the lowest rate of emissions captured was recorded for LDPE and HDPE, and was equivalent to 0.55 % and 0.34%,

323 respectively. The highest mean ATN values recorded corresponded to PET (383), textiles (363) and polystyrene
324 (328). LDPE and paper and cardboard gave the smallest overall ATN values for single waste types, equivalent to
325 46 and 62, respectively.

326 The larger emissions produced by polystyrene compared to HDPE and LDPE is consistent with the rather
327 limited number of investigations that have been completed on the particulate and pollutant emissions from plastics
328 combustion, and could be related to the aromatic chemical structure of polystyrene. For example, Valavanidis et
329 al. (2008) examined the particulate emissions of different plastics including polystyrene, LDPE, HDPE, and PET
330 under controlled laboratory conditions representing open-air combustion. The results demonstrated that
331 polystyrene combustion produced the largest quantity of particulates while HDPE and LDPE yielded the lowest
332 values. In a related study, Shemwell and Levendis (2000) also found that polystyrene combustion produced
333 several times more particulates compared to other polymers.

Table 3 Combustion properties of mixed and single waste types

		Percentage burned (%)	Emissions captured/sample burned (%)	Total combustible mass (%)	Remaining combustible fraction after test (%)	ATN (%)
Mixed waste	National (Mexico) n=11 ATN>500, 6					
	Mean	69	0.94	94	25	400
	sd	5	0.33	159	6	85
	Huejutla n=13 ATN>500, 2					
	Mean	58	0.62	94	35	367
	sd	10	0.23	95	40	103
Single waste	Juchitán n=16 ATN>500, 3					
	Mean	61	0.52	94	33	300
	sd	9.0	0.24	94	42	64
	Paper and cardboard n=23 ATN>500, 0					
	Mean	65	1.6	91	24	62
	sd	12	0.7	91	34	7.7
	Garden waste n=15 ATN>500, 2					
	Mean	57	0.9	94	42	264
	sd	19	0.6	93	74	122
	Textiles n=14 ATN>500, 5					
	Mean	51	1.1	100	49	363
	sd	9	0.4	100	49	114
	LDPE n=17 ATN>500, 0					
	Mean	24	0.3	84	60	46
	sd	6.1	0.1	84	61	6.7
	HDPE n=20 ATN>500, 4					
	Mean	40	0.6	100	59	147
	sd	28	0.5	100	90	19.3
	PET n=17 ATN>500, 9					
	Mean	9.9	1.0	99.6	92.8	383
	sd	9.3	0.3	99.6	103.1	148
	Polystyrene n=14 ATN>500, 4					
	Mean	21	3.0	100	80	328
	sd	7.7	2.2	100	88	162
n = x: number of repetitions ATN>500, x: number of filters marked as "Too Dark" by OT21; sd, standard deviation						

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3.2 Relationship between BC determined by TOA and ATN

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The relationship between the amount of BC detected on the sample filters determined by TOA and transmittance ($\ln I_0/I = \text{ATN}/100$) was non-linear and was represented by a quadratic equation with a high degree of statistical confidence (Figure 5). Several studies have also demonstrated that the rate of change in light absorption through a PM collection filter is non-linear, especially when ATN values are high due to a saturation effect (Ingram and Golden, 1973; Weingartner et al., 2003; Virkkula et al., 2007; Drinovec et al., 2015; Davy et al., 2017). ATN measurements are influenced by a number of factors including multiple light scattering, filter loading and particle scattering effects that produce a non-linear response between ATN and BC (Virkkula et al., 2007; Drinovec et al., 2015; Davy et al., 2017). A quadratic equation therefore provides a good approximation to the underlying processes and is consistent with these other studies (Greiling et al., 2019). Assurance that the

relationship obtained between ATN and BC, measured by TOA, was reliable and consistent was further demonstrated by measurements of two pairs of independent samples with similar ATN, which gave very similar BC results at intermediate and high ATN values (ATN/100 values of approximately 2 and ≈ 4.5 , respectively).

where $y = \text{BC } (\mu\text{g cm}^{-2})$ and $x = \ln(Io/I)$

Figure 4 Relationship between ATN ($\ln(Io/I)$) and TOA-determined BC ($\mu\text{g cm}^{-2}$)

3.3 Emission factors

Emission factors presented in Table 4 were calculated based on data expressed as grams of BC per kg of waste burned following the procedure described in Section 2.7 and the quadratic relationship between ATN and TOA-determined BC shown in Figure 5.

Table 4 Black carbon captured on quartz filters, relative to sample burned, and EF

Type of waste	BC ($\mu\text{g cm}^{-2}$)	Total BC captured (μg)	BC relative to sample burned ($\mu\text{g g}^{-1}$)	BC EF ($\pm\text{sd}$) (g kg^{-1})
Mexico (national scale)	46	418	1,205	1.6 (± 0.3)
Huejutla, Hidalgo, Mexico	39	355	1,000	1.6 (± 0.4)
Juchitán, Oaxaca, Mexico	21	194	804	1.3 (± 0.2)
Paper and cardboard	4	33	15	0.02 (± 0.05)
Garden waste	24	213	276	0.5 (± 0.3)
Textiles	38	349	4,660	9 (± 3.9)
LDPE	3	23	31	0.1 (± 0.1)
HDPE	15	140	67	0.2 (± 0.1)
PET	44	397	1784	46 (± 5.8)
Polystyrene	35	319	10,569	53 (± 34)

The statistical analysis of the EFs for each waste type is shown in Table 4. The highest EFs corresponded to polystyrene (53 g BC kg^{-1}), PET (46 g BC kg^{-1}) and textiles (9 g BC kg^{-1}). Thus, small amounts of these wastes can generate very large BC emissions. For example, polystyrene contributed approximately 44, 15 and 14 % of the total BC emissions, respectively, from the combustion of the National, Huejutla and Juchitán waste compositions; PET, was also a major contributor, equivalent to 34, 63 and 73% of the total overall emissions from these waste mixtures, respectively. The lowest EF corresponded to paper and cardboard (0.02 g BC kg^{-1}), followed

by LDPE (0.1 g BC kg⁻¹). As would be expected, mixed waste containing a larger proportion of PET, polystyrene or textiles had larger EF values for EC overall compared to mixed samples with small amounts of these waste types. Consequently, mixed compositions for Huejutla and National waste presented higher EFs compared to Juchitán, as these samples contained a higher proportion of textiles. However, EFs for mixed waste presented for the case study areas and for Mexico show consistent values ranging from 1.3 to 1.6 g BC kg⁻¹.

Downscaling the experimental measurements of combustion processes may raise questions about the representativeness of the data relative to field conditions. However, the measured EF values for burning waste with the small-scale combustion emissions collection apparatus developed here were within a similar range and in some cases entirely consistent with the limited number of field measurements that have been published in the literature (see Table 5). As would be expected, reported results from measurements of open burning of waste under field conditions show a degree of variation. This could be attributed to differences in the waste composition, although no details are provided in the reported studies, as well as the measurement techniques applied. Indeed, Stockwell et al. (2016) and Jayarathne et al. (2017) used different measuring approaches, but the waste samples investigated were equivalent and in this case their combustion results were relatively similar. The average BC EF measured in the laboratory simulation reported here was approximately 1.5 g kg⁻¹ and was reasonably close to the overall mean value indicated by the available published sources, equivalent to 1.98 g kg⁻¹. Therefore, our results support the application of small-scale, laboratory-based combustion tests to estimate BC EFs from the open burning of waste in agreement with other related research on biomass combustion (Yokelson et al., 2008; McMeeking et al., 2009; Yokelson et al., 2013).

Table 5 Measured BC EFs compared to published values (g kg⁻¹)

BC emission factors (g kg⁻¹ waste)				
Number of tests				
	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 (measured EFs)	Mexico (n=11)	Huejutla (n=13)	Juchitán (n=16)	Average
	1.64	1.58	1.27	1.50
*Three samples were measured but only the average EF was reported.				

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The results showed that, on average, derived mixed waste EFs were approximately five times larger compared to the laboratory measurements of combusted mixed waste samples of equivalent composition. Therefore, a nominal calibration factor of 0.2 was applied to estimated mixed waste EFs (Equation 4) from the single waste, experimental combustion data.

Table 6 Derived EFs (g BC kg⁻¹ of waste burned)

Mixed Waste types	Derived EFs	EF Derived / EF Measured	Average
Mexico	7.45	4.53	4.7
Huejutla, Hidalgo, Mexico	6.57	4.15	
Juchitán, Oaxaca, Mexico	6.83	5.39	

391 The results suggested that the combustion properties of individual materials varied greatly and produced much
 392 larger emissions compared to typical mixtures of different waste streams in practice. Some of the factors that could
 393 influence waste combustion may be related to reduced oxygen availability in more homogenous, single waste
 394 samples compared to mixed waste types since emissions increase from poorly oxygenated, smouldering waste
 395 (Solorzano-Ochoa et al., 2012). This may be explained because homogenous waste samples appeared more
 396 dense and compacted compared to mixed waste samples, resulting in less oxygenated combustion conditions and
 397 thus, higher EFs. Further investigation is required to understand the interaction between different materials during
 398 the combustion process of single and mixed waste types. Nevertheless, a relatively consistent pattern (error = 15
 399 % relative to the mean, Table 6) was observed between the BC EFs determined from the direct measurement of
 400 the combustion emissions from diverse mixed waste samples, representative of different regions in Mexico, and
 401 those derived by summing the emissions of single waste samples constructed to represent equivalent mixtures of
 402 wastes. Therefore, we considered it would be reasonable to apply the calibration factor obtained for the different
 403 waste samples tested for Mexico to derive EFs based on the composition of waste from other global regions
 404 because they represented a similar range of waste compositions found in other developing countries where open
 405 burning of waste is widely practiced.

406 **3.4 Regional BC EFs and global BC emissions from open burning of waste**

407 The experimental EFs were applied to regional waste composition and activity data (waste disposed by burning)
 408 to estimate overall regional BC and BC CO₂ equivalent (BC CO₂Eq) emissions from open burning of waste globally
 409 (Table 7). Black carbon CO₂Eq emissions were calculated using a BC GWP range of 1,200 - 5,100 with a mean
 410 value of 2,200 over a 20-year horizon (Fuglestvedt et al., 2010; Bond et al., 2013). From this calculation, East Asia
 411 and the Pacific (EAP) emit the largest mean amount of BC CO₂Eq from open burning of waste, equivalent to
 412 almost 665 x 10³ t y⁻¹, followed by Latin America and the Caribbean (LCR) which produced approximately 50 % of
 413 the mean BC CO₂Eq emissions from this source compared to EAP, equivalent to 379 x 10³ t y⁻¹. High-income,
 414 OECD countries are responsible for almost half of the solid waste generated globally, however, the amount of BC
 415 CO₂Eq released from open burning is significantly smaller compared to any other region. This is explained
 416 because 98 % of household waste is collected in OECD countries, which is much greater than in other regions
 417 (Hoornweg and Bhada-Tata, 2012) and the open burning of waste in landfills has been eradicated. Nevertheless,
 418 over 10 x 10⁶ t y⁻¹ of waste are burned as a disposal method by households in OECD countries, releasing 10 x 10³
 419 t y⁻¹ of BC. In total, almost 1 x 10⁶ t y⁻¹ of BC are released from open burning of waste globally, equivalent to 1,172
 420 – 4,893 x 10⁶ t y⁻¹ of CO₂Eq. According to the World Bank (World Bank, 2018), the total emission of CO₂Eq in the
 421 world in 2012 was approximately 51,300 x 10⁶ t y⁻¹, therefore, open burning of waste represents approximately 2.3
 422 – 9.7 % of the total global CO₂Eq emissions.

423 **Table 7 BC, CH₄ and CO₂Eq emissions for different regions of the world (t y⁻¹ unless otherwise stated)**

							^(a) EF CH ₄ (MTCO ₂ Eq Wet t ⁻¹)				
							1.59	0.63	2.3		
	^(b) Waste burned (t x 10 ⁶ y ⁻¹)	BC Emission Factors (kg t ⁻¹)	BC Emissions			^(d) Food waste (t x 10 ⁶ y ⁻¹)	Garden waste (t x 10 ⁶ y ⁻¹)	Paper & cardboard (t x 10 ⁶ y ⁻¹)	^(e) CH ₄ CO ₂ Eq emissions (t x 10 ⁶ y ⁻¹)	BC CO ₂ Eq / CH ₄ CO ₂ Eq Lower – Upper, Mean	
			BC (t x 10 ⁶ y ⁻¹)	^(c) BC (CO ₂ Eq) (t x 10 ⁶ y ⁻¹)							
				Lower	Mean	Upper					
Africa Region (AFR)	107.3	1.15	0.124	149	272	632	23.1	15.1	9.7	68.3	2.2 - 9.3, 4.0
East Asia & Pacific (EAP)	396.6	0.76	0.302	362	665	1540	99.1	47.6	39.7	278.5	1.3 - 5.5, 2.4
Eastern & Central Asia (ECA)	99.2	1.08	0.107	128	235	546	9.7	27.2	13.9	64.4	2.0 - 8.5, 3.6
South Asia (SAR)	153.8	1.33	0.204	245	449	1,040	31.6	13.6	6.2	72.9	3.4 - 14.3, 6.2
Middle East & North Africa (MENA)	61.5	0.96	0.059	71	130	301	16.9	3.7	8.6	49.0	1.4 - 6.1, 2.6
Latin America & the Caribbean (LAC)	143.7	1.20	0.172	206	379	877	28.0	21.5	23.0	110.9	1.9 - 7.9 - 3.4
Organisation for Economic Co-operation and Development (OECD)	10.1	0.97	0.010	12	22	51	0.6	1.6	3.2	9.4	11.3 - 2.3, 5.4
WORLD	972.2	-	0.977	1,172	2,150	4,893	209.0	130.3	104.2	653.3	1.9 - 8.1, 3.5 (Global average)

a) Emission factors for CH₄ in metric tonnes of CO₂ equivalent per wet tonne of waste are taken from the Waste Reduction Model (WARm) (US EPA, 2015).
b) Source: Wiedinmyer et al. (2014).
c) Global warming potential (GWP) for BC in a 20 year horizon is 1,200 – 5,100 with a mean value of 2,200 (Fuglestvedt et al., 2010; Bond et al., 2013).
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.
General Note: Waste composition was determined using data from Hoornweg and Bhada-Tata (2012) and by the sources provided in the supplementary material.

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425 Methane emissions from decomposition of biodegradable waste from landfill and dumpsite disposal is
426 recognised as one of the major anthropogenic sources of the gas impacting climate change and has therefore
427 received considerable attention and has led to international regulations and controls on the disposal of
428 biodegradable waste in landfill (European Union, 1999). Therefore, to place the relative importance and impact of
429 BC from uncontrolled open burning into context we have compared the CO₂Eq from both sources, by assuming
430 the equivalent amount of waste that is currently disposed by open burning is disposed of in landfill/dumpsites.
431 Therefore, CH₄ CO₂Eq emissions for waste that is currently burned were calculated for different major international
432 regions and the results showed that BC CO₂Eq from burning were on average between 2 and 8 (mean = 3.5)
433 times larger than CH₄ CO₂Eq emissions from disposing of equivalent amounts of biodegradable waste in
434 landfill/dumpsites (Table 6). The scale of the impacts varied between the regions and burning may produce twice
435 the amount of CO₂Eq emissions compared to disposing of the equivalent amount of waste by landfilling in the
436 OECD countries compared to the South Asia region (SAR), where burning waste may produce 6 times more
437 CO₂Eq emissions than disposing of the waste in landfill/dumpsites. The differences in the magnitude of the impact
438 between open burning and landfilling on the CO₂Eq emissions from BC or CH₄, respectively, were explained due
439 to the relative composition of combustible and biodegradable materials in the waste streams from these regions.

440 **4 Conclusions**

441 We have developed BC EFs relevant to the open, uncontrolled burning of waste based on a small-scale,
442 laboratory experimental combustion system. Black carbon EFs from individual waste streams were measured and
443 the values were combined in proportion to representative waste compositions to provide a derived EF value; the
444 derived results were compared to the measured EFs for these contrasting waste mixtures. Importantly, the EFs
445 obtained for mixed waste compositions were similar to published field data on BC emissions from uncontrolled

burning of waste and, thus, provided independent validation of the approach. Combustion emissions from burning individual types of waste separately are typically much larger than is expected from complex mixtures of disposed wastes. This is due to factors such as the increased density of the waste, which inhibits the transfer of oxygen, therefore reducing overall combustion efficiency. We found that the differences in derived BC EFs obtained from combined single wastes and EF values measured from combusting the mixed waste samples were consistent, and, on average, the ratio was equivalent to 0.2. Therefore, (a) given that the results from the combustion of contrasting mixed waste samples were comparable to field data, and (b) the ratios of the differences between individual waste EFs and for the equivalent combined mixtures were consistent, we applied this approach to estimate BC EFs for different regions of the world based on the characteristic waste composition from these areas.

The fraction sizes of different types of plastic in the waste streams in Mexico and other developing regions of the world are relatively small compared to organic wastes, and typically are in the range 5 - 10%, although they can be smaller than this, for example, the polystyrene content of waste in the Huejutla and Juchitán regions was <2%. Nevertheless, the large BC EFs observed particularly for PET and polystyrene, compared to the other plastic and organic waste materials, mean that these plastics are the main contributors to the overall BC emissions from open burning, together representing up to approximately 90% of the total emitted BC. Therefore, introducing measures through public education and targeted waste segregation and collection of these specific streams would have a major and direct benefit in reducing BC emissions from uncontrolled open burning of waste.

The results demonstrated that the contribution of uncontrolled open burning of waste to global BC emissions represented between 2 and 10 % of total CO₂Eq emissions from all sources. Indeed, global BC CO₂Eq emissions from burning waste may be approximately between 2 and 8 times larger compared to CH₄ CO₂Eq 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, as well as minimising its contribution to climate change. The results presented here suggest the uncontrolled burning of waste is a much larger contributor to climate change compared to the CH₄ emissions from landfill or dumpsite 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.

5 Recommendations for further research

The combustion process and the character of the associated emissions are strongly dependent upon the physical conditions of the combusting material. Mixed waste samples are physically complex materials and can differ in moisture content, particle size distribution and density. It would therefore be valuable to consider the impacts of these factors on the extent of BC emissions from open burning. This would require an alternative experimental approach to that described here, which would be capable of handling larger sample sizes necessary to provide representative physical conditions, and with more sophisticated combustion control and emissions capture systems. Altitude is another factor that affects the availability of oxygen and the combustion process and could be considered in a refined BC EF set for open burning of waste.

483 In this work we have attempted to derive EF values for different mixed waste streams representative of the
484 developing regions of the world based on controlled, laboratory-based combustion tests of single waste materials.
485 We supported this assumption based on the consistent pattern found between the burning properties of single and
486 mixed waste samples from Mexico, which were representative of the waste compositions found in other developing
487 areas. However, the model could be improved by gaining a better understanding of the interactions between
488 different waste types during the combustion process. Measuring specific EFs for the mixed waste compositions
489 representative of other regional areas would also provide further validation of the approach.

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499 7 References

- 500 Ahmed, T., Dutkiewicz, V. A., Shareef, A., Tuncel, G., Tuncel, S. & Husain, L. 2009. Measurement of black carbon
501 (BC) by an optical method and a thermal-optical method: Intercomparison for four sites. *Atmospheric*
502 *Environment*, 43, 6305-6311.
- 503 Aurell, J., Mitchell, W., Chirayath, V., Jonsson, J., Tabor, D. & Gullett, B. 2017. Field determination of multipollutant,
504 open area combustion source emission factors with a hexacopter unmanned aerial vehicle. *Atmospheric*
505 *Environment*, 166, 433-440.
- 506 Institution 2007. Chapter 10: Waste management. In: Gregory, R. & (Eds.), R. T. M. S. (eds.) *Contribution of*
507 *Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change,*
508 2007. Cambridge, United Kingdom and New York, NY, USA.: IPCC. Available at:
509 <https://www.ipcc.ch/pdf/assessment-report/ar4/wg3/ar4-wg3-chapter10.pdf>. Accessed on October 4,
510 2017.
- 511 Bond, T. C., Doherty, S. J., Fahey, D. W., Forster, P. M., Berntsen, T., DeAngelo, B. J., Flanner, M. G., Ghan, S.,
512 Kärcher, B., Koch, D., Kinne, S., Kondo, Y., Quinn, P. K., Sarofim, M. C., Schultz, M. G., Schulz, M.,
513 Venkataraman, C., Zhang, H., Zhang, S., Bellouin, N., Guttikunda, S. K., Hopke, P. K., Jacobson, M. Z.,
514 Kaiser, J. W., Klimont, Z., Lohmann, U., Schwarz, J. P., Shindell, D., Storelvmo, T., Warren, S. G. &
515 Zender, C. S. 2013. Bounding the role of black carbon in the climate system: A scientific assessment.
516 *Journal of Geophysical Research: Atmospheres*, 118, 5380-5552.
- 517 Briggs, N. L. & Long, C. M. 2016. Critical review of black carbon and elemental carbon source apportionment in
518 Europe and the United States. *Atmospheric Environment*, 144, 409-427.
- 519 Cavalli, F., Viana, M., Yttri, K. E., Genberg, J. & Putaud, J. P. 2010. Toward a standardised thermal-optical protocol
520 for measuring atmospheric organic and elemental carbon- the EUSAAR protocol. *Atmospheric*
521 *Measurement Techniques*, 3, 79-89.
- 522 Christian, T. J., Yokelson, R. J., Cardenas, B., Molina, L. T., Engling, G. & Hsu, S. C. 2010. Trace gas and particle
523 emissions from domestic and industrial biofuel use and garbage burning in central Mexico. *Atmospheric*
524 *Chemistry and Physics*, 10, 565-584.

525 Davy, P. M., Tremper, A. H., Nicolosi, E. M. G., Quincey, P. & Fuller, G. W. 2017. Estimating particulate black
526 carbon concentrations using two offline light absorption methods applied to four types of filter media.
527 *Atmospheric Environment*, 152, 24-33.

528 Drinovec, L., Močnik, G., Zotter, P., Prévôt, A. S. H., Ruckstuhl, C., Coz, E., Rupakheti, M., Sciare, J., Müller, T.,
529 Wiedensohler, A. & Hansen, A. D. A. 2015. The "dual-spot" Aethalometer: an improved measurement of
530 aerosol black carbon with real-time loading compensation. *Atmospheric Measurement Techniques*, 8,
531 1965-1979.

532 European Union 1999. Landfill directive 1999/31/EC. Official Journal of the European Communities, Luxembourg.
533 Available at: <https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX%3A31999L0031>. Accessed on
534 August 8, 2018.

535 Fuglestad, J. S., Shine, K. P., Berntsen, T., Cook, J., Lee, D. S., Stenke, A., Skeie, R. B., Velders, G. J. M. &
536 Waitz, I. A. 2010. Transport impacts on atmosphere and climate: Metrics. *Atmospheric Environment*, 44,
537 4648-4677.

538 Greiling, M., Drinovec, L., Močnik, G. & Kasper-Giebl, A. 2019. Evaluation of measurements of light transmission
539 for the determination of black carbon on filters from different station types. *Atmospheric Environment*, 198,
540 1-11.

541 Hansen, A. D. A. 2005. The Aethalometer. Section The Aethalometer. Berkeley, California, USA. Available at:
542 https://www.psi.ch/catcos/ProjectDetailCatcosOperationsEN/Aethalometer_book_2005.07.02.pdf.
543 Accessed on October 26, 2017.

544 Hansen, A. D. A., Rosen, H. & Novakov, T. 1984. The Aethalometer - An instruments for the real-time
545 measurement of optical absorption by aerosol particles. *The Science of the Total Environment*, 36, 191-
546 196.

547 Hoornweg, D. & Bhada-Tata, P. 2012. What a waste : a global review of solid waste management. Washington
548 D.C. Available at: http://siteresources.worldbank.org/INTURBANDEVELOPMENT/Resources/336387-1334852610766/What_a_Waste2012_Final.pdf. Accessed on June 11th, 2016.

550 INECC 2012. Diagnóstico Básico para la Gestión Integral de los Residuos 2012. Versión Extensa. Mexico.
551 Available at: http://www.inecc.gob.mx/descargas/dgcnica/diagnostico_basico_extenso_2012.pdf.
552 Accessed on May 9, 2017.

553 Ingram, W. T. & Golden, J. 1973. Smoke Curve Calibration. *Journal of the Air Pollution Control Association*, 23,
554 110-115.

555 IPCC 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth
556 Assessment Report of the Intergovernmental Panel on Climate Change. Geneva, Switzerland. Available
557 at: http://ar5-syr.ipcc.ch/ipcc/resources/pdf/IPCC_SynthesisReport.pdf. Accessed on October 12,
558 2017.

559 ISO 2008. 3451-1:2008: Determination of ash. 3451-1. Physical-chemical properties Committee. International
560 Organization for Standardization. Available at: <https://www.iso.org/standard/41663.html>. Accessed on
561 March 2, 2016.

562 Janssen, N. A., Gerlofs-Nijland, M. E., Lanki, T., Salonen, R. O., Cassee, F., Hoek, G., Fischer, P., Brunekreef, B.
563 & Krzyzanowski, M. 2012. Health effects of Black Carbon. Copenhagen. Available at:
564 http://www.euro.who.int/_data/assets/pdf_file/0004/162535/e96541.pdf. Accessed on July 5, 2016.

565 Jayarathne, T., Stockwell, C. E., Bhave, P. V., Praveen, P. S., Rathnayake, C. M., Islam, M. R., Panday, A. K.,
566 Adhikari, S., Maharjan, R., Goetz, J. D., DeCarlo, P. F., Saikawa, E., Yokelson, R. J. & Stone, E. A. 2017.
567 Nepal Ambient Monitoring and Source Testing Experiment (NAMaSTE): Emissions of particulate matter
568 from wood and dung cooking fires, garbage and crop residue burning, brick kilns, and other sources.
569 *Atmospheric Chemistry and Physics Discussions*, 2017, 1-51.

570 Karanasiou, A., Minguillón, M. C., Viana, M., Alastuey, A., Putaud, J.-P., Maenhaut, W., Panteliadis, P., Mocnik,
571 G., Favez, O. & Kuhlbusch, T. A. J. 2015. Thermal-optical analysis for the measurement of elemental
572 carbon (EC) and organic carbon (OC) in ambient air a literature review. *Atmospheric Measurement
573 Techniques*, 8, 9649-9712.

574 Lemieux, P. M., Lutes, C. C. & Santoianni, D. A. 2004. Emissions of organic air toxics from open burning: a
575 comprehensive review. *Progress in Energy and Combustion Science*, 30, 1-32.

Lioussé, C., Cachier, H. & Jennings, S. G. 1993. Optical and thermal measurements of black carbon aerosol content in different environments: Variation of the specific attenuation cross-section, σ . *Atmospheric Environment. Part A. General Topics*, 27, 1203-1211.

Long, C. M., Nascarella, M. A. & Valberg, P. A. 2013. Carbon black vs. black carbon and other airborne materials containing elemental carbon: Physical and chemical distinctions. *Environmental Pollution*, 181, 271-286.

Magee Scientific 2015. SootScan™ Model OT21 Transmissometer Product Specifications. Aerosol d.o.o. Section SootScan™ Model OT21 Transmissometer Product Specifications. Ljubljana, Slovenia. Available at: http://group.mageesci.com/images/sampled/ATAdata/brochures/OT21/Magee_Scientific_SootScan_OT21_Transmissometer_brochure_sheet_A4.pdf. Accessed on November 1, 2016.

McMeeking, G. R., Kreidenweis, S. M., Baker, S., Carrico, C. M., Chow, J. C., Jr., J. L. C., Hao, W. M., Holden, A. S., Kirchstetter, T. W., Malm, W. C., Moosmüller, H., Sullivan, A. P. & Wold, C. E. 2009. Emissions of trace gases and aerosols during the open combustion of biomass in the laboratory. *Journal of Geophysical Research: Atmospheres*, 114

Ortiz-de-Zárate, Ezcurra, A., Lacaux, J. P. & Dinh, P. V. 2000. Emission factor estimates of cereal waste burning in Spain. *Atmospheric Environment*, 34, 3183-3193.

Park, Y. K., Wooram, K. & Young Min, J. 2013. Release of Harmful Air Pollutants from Open Burning of Domestic Municipal Solid Wastes in a Metropolitan Area of Korea. *Aerosol and Air Quality Research*,

Petzold, A., Kopp, C. & Niessner, R. 1997. The dependence of the specific attenuation cross-section on black carbon mass fraction and particle size. *Atmospheric Environment*, 31, 661-672.

Petzold, A., Ogren, J. A., Fiebig, M., Laj, P., Li, S. M., Baltensperger, U., Holzer-Popp, T., Kinne, S., Pappalardo, G., Sugimoto, N., Wehrli, C., Wiedensohler, A. & Zhang, X. Y. 2013. Recommendations for reporting "black carbon" measurements. *Atmospheric Chemistry and Physics*, 13, 8365-8379.

Reyna-Bensusan, N. 2019. *The Impact of Black Carbon Emissions from Open Burning of Solid Waste*. PhD, Imperial College London.

Reyna-Bensusan, N., Wilson, D. C. & Smith, S. R. 2018. Uncontrolled burning of solid waste by households in Mexico is a significant contributor to climate change in the country. *Environmental Research*, 163, 280-288.

SEMARNAT 2013a. Sistema Nacional de Información Ambiental y de Recursos Naturales. Módulo de consulta temática, dimensión ambiental, generación de residuos sólidos urbanos 2012. Available at: http://dgeiawf.semarnat.gob.mx:8080/approot/dgeia_mce/html/mce_index.html?De=BADESNIARN. Accessed on December 15, 2016.

SEMARNAT 2013b. Informe de la Situación del Medio Ambiente en México: Compendio de Estadísticas Ambientales. Indicadores Clave y de Desempeño Ambiental (Capítulo 7: Residuos). Ciudad de México. Available at: https://www.google.co.uk/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=0ahUKEwjrzYbU0-QAhUqAsAKHZC-Aj0QFggaMAA&url=http%3A%2F%2Fapps1.semarnat.gob.mx%2Fdgeia%2Finforme_12%2Fpdf%2Finforme_2012.pdf&usq=AFQjCNGo2-A7ayGBzgjW4XxyPWm349YFQ&sig2=W74qwqeL1uqaU6szZiUe-g. Accessed on December 12, 2016.

Shemwell, B. E. & Levendis, Y. A. 2000. Particulates Generated from Combustion of Polymers (Plastics). *Journal of the Air & Waste Management Association*, 50:1, 94-102.

Shreve, B., Thiex, N. & Wolf, M. 2006. National Forage Testing Association Reference Method: Dry Matter by Oven Drying. Omaha, NB, USA. Available at: <http://www.foragetesting.org/files/NFTARReferenceMethodDM.pdf>. Accessed on January 31, 2016.

Sluiter, A., Hames, B., Ruiz, R., Scarlata, C., Sluiter, J. & Templeton, D. 2008. Determination of Ash in Biomass.

Solorzano-Ochoa, G., de la Rosa, D. A., Maiz-Laralde, P., Gullett, B. K., Tabor, D. G., Touati, A., Wyrzykowska-Ceradini, B., Fiedler, H., Abel, T. & Carroll, W. F. 2012. Open burning of household waste: Effect of experimental condition on combustion quality and emission of PCDD, PCDF and PCB. *Chemosphere*, 87, 1003-1008.

625 Stockwell, C. E., Christian, T. J., Goetz, J. D., Jayarathne, T., Bhawe, P. V., Praveen, P. S., Adhikari, S., Maharjan,
626 R., DeCarlo, P. F., Stone, E. A., Saikawa, E., Blake, D. R., Simpson, I. J., Yokelson, R. J. & Panday, A.
627 K. 2016. Nepal Ambient Monitoring and Source Testing Experiment (NAMASte): emissions of trace gases
628 and light-absorbing carbon from wood and dung cooking fires, garbage and crop residue burning, brick
629 kilns, and other sources. *Atmospheric Chemistry and Physics*, 16, 11043-11081.

630 Tissari, J., Lyyrnen, J., Hytnen, K., Sippula, O., Tapper, U., Frey, A., Saarnio, K., Pennanen, A. S., Hillamo, R.,
631 Salonen, R. O., Hirvonen, M. R. & Jokiniemi, J. 2008. Fine particle and gaseous emissions from normal
632 and smouldering wood combustion in a conventional masonry heater. *Atmospheric Environment*, 42,
633 7862-7873.

634 UNEP 2011. Integrated Assessment of Black Carbon and Tropospheric Ozone: Summary for Decision Makers.
635 Available at: https://www.wmo.int/pages/prog/arep/gaw/documents/BlackCarbon_SDM.pdf. Accessed on
636 December 3, 2018.

637 UNEP & ISWA 2015. Global Waste Management Outlook United Nations Development Programme. Available at:
638 [http://www.unep.org/ietc/InformationResources/Events/](http://www.unep.org/ietc/InformationResources/Events/GlobalWasteManagementOutlookGWMO/tabid/106373/Default.aspx)
639 [GlobalWasteManagementOutlookGWMO/tabid/106373/Default.aspx](http://www.unep.org/ietc/InformationResources/Events/GlobalWasteManagementOutlookGWMO/tabid/106373/Default.aspx). Accessed on July 29, 2016.

640 US EPA 2015. Waste Reduction Model (WARM), Version 13: Landfilling. Washington DC. Available at:
641 <https://www3.epa.gov/epawaste/conserve/tools/warm/pdfs/Landfilling.pdf>. Accessed on May 4, 2017.

642 Valavanidis, A., Iliopoulos, N., Gotsis, G. & Fiotakis, K. 2008. Persistent free radicals, heavy metals and PAHs
643 generated in particulate soot emissions and residue ash from controlled combustion of common types of
644 plastic. *Journal of Hazardous Materials*, 156, 277–284.

645 Virkkula, A., Mkel, T., Hillamo, R., Yli-Tuomi, T., Hirsikko, A., Hmeri, K. & Koponen, I. K. 2007. A Simple
646 Procedure for Correcting Loading Effects of Aethalometer Data. *Journal of the Air & Waste Management*
647 *Association*, 57, 1214-1222.

648 Watson, J. G., Chow, J. C. & Chen, L. W. A. 2005. Summary of organic and elemental carbon/black carbon
649 analysis methods and intercomparisons. *Aerosol and Air Quality Research*, 5, 69–102.

650 Weingartner, E., Saathoff, H., Schnaiter, M., Streit, N., Bitnar, B. & Baltensperger, U. 2003. Absorption of light by
651 soot particles: determination of the absorption coefficient by means of aethalometers. *Journal of Aerosol*
652 *Science*, 34, 1445-1463.

653 Wiedinmyer, C., Yokelson, R. J. & Gullett, B. K. 2014. Global emissions of trace gases, particulate matter, and
654 hazardous air pollutants from open burning of domestic waste. *Environment, Science and Technology*, 48,
655 9523-9530.

656 World Bank. 2018. *Total greenhouse gas emissions*. Available:
657 <https://data.worldbank.org/indicator/EN.ATM.GHGT.KT.CE?end=2012&start=2009> Accessed on
658 February 22 2018.

659 Yelverton, T. L. B., Hays, M. D., Gullett, B. K. & Linak, W. P. 2014. Black Carbon Measurements of Flame-
660 Generated Soot as Determined by Optical, Thermal-Optical, Direct Absorption, and Laser Incandescence
661 Methods. *Environmental Engineering Science*, 31, 209-215.

662 Yokelson, R. J., Burling, I. R., Gilman, J. B., Warneke, C., Stockwell, C. E., de Gouw, J., Akagi, S. K., Urbanski,
663 S. P., Veres, P., Roberts, J. M., Kuster, W. C., Reardon, J., Griffith, D. W. T., Johnson, T. J., Hosseini, S.,
664 Miller, J. W., Cocker III, D. R., Jung, H. & Weise, D. R. 2013. Coupling field and laboratory measurements
665 to estimate the emission factors of identified and unidentified trace gases for prescribed fires. *Atmos.*
666 *Chem. Phys.*, 13, 89-116.

667 Yokelson, R. J., Christian, T. J., Karl, T. G. & Guenther, A. 2008. The tropical forest and fire emissions experiment-
668 laboratory fire measurements and synthesis of campaign data. *Atmospheric Chemistry and Physics*, 8,
669 3509–3527.

670 Zhang, J. & Morawska, L. 2002. Combustion sources of particles: 2. Emission factors and measurement methods.
671 *Chemosphere*, 49, 1059-1074.

672