Emission factors for PM$_{10}$ and PAHs from illegal burning of different types of municipal waste in households

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Abstract. It is a common practice in the developing countries and in some regions of Europe that solid wastes generated in the households (e.g. plastic beverage packaging and other plastic wastes, textile wastes, fibreboards, furniture, tyres, and coloured paper waste) are burned in wood- or coal-fired stoves during the winter months. In Europe, the types and volume of municipal waste burned in households is virtually unknown because these activities are illegal and not recorded, with the exception of a few media reports or court cases. Even though particulate emissions from illegal waste burning pose an unprecedented hazard to human health due to the combination of excessive emission factors (EFs) and uncontrolled chemical composition, there is scarce information on the specific emission factors for PM$_{10}$ and polycyclic aromatic hydrocarbons (PAHs) in the scientific literature. In this work, controlled combustion tests were performed with 12 different types of municipal solid waste and particulate emissions were measured and collected for chemical analysis. Absolute emission factors for PM$_{10}$ and PAHs as well as the benzo(a)pyrene toxicity equivalent of the latter are reported for the first time for the indoor combustion of 12 common types of municipal solid waste that are frequently burned in households worldwide.

1 Introduction

In developing countries more than three billion people use solid fuels (wood, coal or agricultural waste) as the main source of household energy (Anenberg et al., 2013), the latter is extensively used in India (Saud et al., 2011). It is well known that fossil fuel combustion and biomass burning are the two most important sources of fine particulate matter in the atmosphere (Simoneit et al., 2002). Karagulian et al. (2015) estimated that domestic fuel burning (wood, coal and gas) might contribute up to 32 % of PM$_{2.5}$ emissions, and up to 45 % of PM$_{10}$ emissions in Central and Eastern Europe (globally 20 % and 15 %, respectively). In Europe solid fuels (primarily wood but in some countries also coal) are extensively used for home heating. Residential wood combustion was found to be the main emission source of fine particles all over Europe in winter (Tissari et al., 2008; Puxbaum et

al., 2007; Gelencsér et al., 2007; Marmureanu et al., 2020). Emissions from wood burning were estimated to contribute to about 30 % of the $PM_{2.5}$ fraction in Portugal (Goncalves et al., 2012). In Lombardia somewhat lower contributions (5−25 % to PM_{10}) were estimated (Pastorello et al., 2011), even in the city of Milan

40 (Piazzalunga et al., 2011). Based on recent studies fine particles from wood combustion may cause severe health effects such as lung cancer, chronic lung and heart diseases (Rafael et al., 2015; Bai et al., 2015), contribute to visibility reduction (Pipal and Satsangi, 2015), and even modify synoptic conditions (Tiwari et al., 2015).

On top of the significant emissions from the burning of solid fuels in households there is another existing yet largely unknown source of atmospheric pollution, the illegal burning of municipal solid waste in households in

45 several countries of the world, even in Europe. In Hungary, two independent polls have recently revealed that 2−10 % of the population burn their wastes in stoves on a regular basis (Kantar Hoffman LTD., 2017; Századvég Foundation, 2018). The major types of waste burned were treated wood (furniture, OSB, hardboard, painted wood, plywood), clothes, plastics, tyres, and used oil. In many villages in Romania the burning of household waste is an everyday practice due to the lack of organised waste collection system. People who heat their homes

50 with wood- or coal-fired stoves usually also burn plastics, tyres, clothes, treated wood, and virtually all types of combustible waste (Autogreen, 2010). Burning any type of waste poses excessive risks to the health of people living in those areas since a plethora of toxic, carcinogenic or mutagenic compounds are emitted in immense quantities compared to the burning of authorised solid fuels such as dry fuel wood or high-quality coal. Since burning any kinds of municipal waste in households is strictly prohibited all over Europe, understandably such

55 emissions are neither specified nor regulated. Being an illegal activity, even its magnitude is mostly unknown apart from a few public reports of NGOs, media coverages, and documented court cases. More interestingly, there has been no single systematic study on the EFs for burning abundant types of solid waste in households in the scientific literature. There are only a small number of scientific papers on the burning of different types of (mostly plastic) waste with the specific focus of finding organic tracers for their tracking in atmospheric

60 particulate matter. This is surprising in the light of the fact that in developing countries solid waste burning is a known and important source of particulate pollution. Among, there are only a handful of studies that report EFs of $PM_{2.5}$ for the open burning of municipal mixed solid waste, in the order of 10 g kg^{-1} (Christian et al., 2010; Park et al., 2013; Jayarathne et al., 2018). Polycyclic aromatic hydrocarbons (PAHs) are among the most hazardous combustion products due to their carcinogenic and mutagenic nature. Among them 5-ring PAHs such

65 as benzo(a)pyrene, benzofluoranthenes, dibenzo(a,h)antracene are known as highly carcinogenic compounds. The EFs of PAHs may vary for different waste types, but such data are scarcely available in the scientific literature though absolute EFs would be essential to assess the impact of residential waste burning on air quality. To the best of our knowledge there has been only a single study reporting $PM_{2.5}$ and PAHs emission factors for co-combustion of PE and PET waste with beech logs in a commercial boiler (Tomsej et al., 2018). In this study

70 EFs for total PAH and PM_{10} were determined for 12 waste types and compared to those of wood burning

2 Material and methods

2.1 Waste samples for combustion tests

The types of solid waste specimens for the combustion tests were selected based on their abundance in households and available information on illegal waste burning practices in Hungary. Polyethylene terephthalate

75   (PET), Polystyrene (PS), Polypropylene (PP), Polyethylene (PE), Polyvinyl chloride (PVC) Polyurethane (PU),
Acrylonitrile Butadiene Styrene (ABS), tyre (TR), Oriented Strand Board (OSB), laminated Melamine Low-
Density Fibreboard (LDF), rag (RAG), paper (PAP), and firewood (WOOD) were selected for waste combustion
tests (Figure 1).

80   **Figure 1 Solid waste specimens prepared for combustion tests.**

The PET waste samples were prepared from beverage plastic bottles (volume 1.5 and 2 L) without cap and label.
The PS waste samples were hard cups/pots of dairy products (yoghurt and pudding) without aluminium foil cap,
and pieces of expanded polystyrene (EPS) insulation board (80 mm × ~100 mm × ~100 mm). The hard and
expanded PS waste sample specimens were burned separately. The PP waste sample specimens were the mixture
85   of plastic cups/pots of dairy products (sour cream and pudding) without paper labels and aluminium foils and
quartered plastic tray of meat. The PE waste specimens were prepared from the mixture of high and low density
polyethylene (HDPE and LDPE). The HDPE and LDPE fractions consisted of plastic caps of beverage bottles
and pieces of various foils and plastic bags, respectively. The PVC waste samples consisted of soft packaging,
small pieces of vinyl flooring, and hard plastic water pipe. The PU waste samples consisted of pieces of packing
90   sponge (average size: 120 mm × 100 mm × 15 mm). The ABS samples were shredded pieces of stands of
computer monitors (average size: 40 mm × 40 mm × 20 mm). The OSB samples consisted of slices of OSB
material of different brands (average size: 120 mm × 100 mm × 20 mm). The LDF (fibreboard) samples were
pieces (average size: 130 mm × 100 mm × 20 mm) of different fibreboards including coloured laminated coating
and plastic border. The tyre (TR) sample specimens consisted of pieces of a new and old tyre of a van and a
95   passenger car, respectively (average size: 80 mm × 40 mm × 15 mm). The textile (RAG) samples consisted of a
mixture of cotton, polyester, and polyamide fabrics from different clothes. Ball-shaped specimens (average
weight: ~70 g) of two types of paper (PAP) samples (colourful glossy coated paper and uncoated paper from
advertising flyers and newspaper) were burned separately. The fuel wood (WOOD) samples consisted of pieces
of logs (average weight: ~130 g) of Turkey oak (*Quercus cerris*) and black locust (*Robinia pseudoacacia*). All
100   sample specimens were weighted with an analytical balance before the combustion tests. Authorisation for
conducting controlled waste burning tests was granted by the Department of Environmental Protection and
Nature Conversation of Veszprém District.

2.2 Experimental conditions of the combustion tests

Combustion tests were carried out in a commercially available cast iron stove (type: Servant S114, heating
105   power: 5 kW). The stove was heated up with smouldering charcoal for ~1 hour prior to the start of combustion
tests which produced very low particulate emission baseline throughout the duration of the measurements.
During each combustion test 1–10 aliquots of solid waste were burned depending on the emission characteristics
of the given waste type. The mass of each sample aliquot was measured with an analytical balance and was
recorded. The air supply ratio of the stove could be set to low or high. Each type of waste (except paper) was
110   burned at high, combined and low excess of air supply ratios resulting in different temperatures and conditions
inside the oven. The temperature of flue gas was measured before and after each measurement by a K-type
thermocouple thermometer (maximum temperature 1000 °C; Testo 925) in the stack 11 cm above the exhaust
opening of the stove, respectively. Between the different sample runs the stove and the stack were heated up to

above 700 °C for a minimum of 10 minutes to minimise cross-contamination between combustion tests with
115 different waste types. The temperature of the flue gas served as an indicator for the experimental conditions of
each combustion test run. The temperature values of the flue gas during the experiments were stable as the mean
temperature values were 299 °C (SD: 11 °C) and 233 °C (SD: 10 °C) at high and low air supply ratio,
respectively (see Fig. S1).

The mixing ratios of CO_2 and O_2 in the flue gas were measured with a CO_2–O_2 analyser (Servomex) at the end of
120 the stack. At the end of the stack a small amount of the flue gas was introduced through a brass tubing (ø 5 mm)
into a dilution unit of a volume of 80 L and was diluted with ambient air. The concentration of CO_2 in the diluted
flue gas was also monitored with a SensAir CO_2 analyser at the inlet of the filter sampling head. The dilution
ratio was determined from the ratio of the measured concentrations of CO_2 at the end of the stack and in the
dilution unit (taking into account the ambient CO_2 concentration as well). The mean dilution ratio was about 81.2
125 (SD: 12) (see Fig. S2) which is independent of the air supply ratios but may depend on the progress of the
burning process.

Figure 2 shows the variation of the CO_2 and O_2 mixing ratios at the end of the stack and the CO_2 mixing ratio in
the diluted flue gas for a PS sample. Each peak corresponds to the burning of a single aliquot of the solid waste.
The collections of each PM_{10} aerosol sample were started after reaching stabile baselines of CO_2 and O_2
130 concentration values and were finished after the return of stable baselines at the end of combustion of all aliquots
of solid wastes.

**Figure 2 Example (PS-F9 sample) for the variations of the measured CO_2 and O_2 mixing ratios at the end of the stack
and the CO_2 mixing ratio in the diluted flue gas during combustions tests.**

135 PM_{10} aerosol samples were collected on quartz filters of 150 mm in diameter (Advantec QR-100 quartz fiber,
binder free) with a high-volume aerosol sampler (flow rate 32 m^3 h^{-1}; Kalman System Co., Hungary) at the
dilution unit. Blank samples were also collected for each waste type. The quartz filters were conditioned at a
temperature of 20 ± 1 °C and relative humidity (RH) of 45−50 % for three days and were weighed in an isolated
weighing room before and after the aerosol samplings according to the European standard (MSZ EN 12341:
140 2014). The parameters were measured and collected by a data acquisition system. The weighted filters were
stored in glass petri dishes (preheated at 450 °C) prior to sampling, whereas the exposed filters were stored in the
freezer in glass petri dishes wrapped into aluminium foil until conditioning and measurements.

Table S1 lists the key parameters of the combustion tests, including the type and mass of sample specimens, the
air supply settings, the number of test burns (by air supply settings), and the measured blank corrected PM_{10}
145 mass on each filter. The mass of the waste sample specimens burned was optimised in preliminary tests to yield
PM_{10} concentrations of about the same magnitude in each combustion test. Since different waste types yielded
vastly different particulate emissions upon burning (e.g. PS, PP, PE, PVC, PU, ABS, and TR were superemitters
compared to PET, OSB, LDF, RAG, PAP and WOOD), this step was necessary to avoid massive overloading of
the filters and the measuring instrumentation. Thus the measurements were comparable and the measured mass
150 of PM_{10} on filters was kept in the range of 5.4 mg and 37.2 mg for all combustion tests.

2.3 Analysis of polycyclic aromatic hydrocarbons in the filter samples

The amount of PAHs in the filter samples was determined by analysing a filter spot with a diameter of 1.4 cm.
First PAHs were extracted with 4.5 mL hexane in an ultrasonic bath for 15 minutes then the extract was filtered

through ashless quantitative (Grade 44, Whatman, UK) filter paper and cleaned on 0.5 g Florisil adsorbent. The

155　cleaned extract was gently evaporated to dryness and re-dissolved in 1.0 mL acetonitrile. The concentration of

15 PAHs out of the 16 EPA priority pollutant PAHs was determined in the final aliquot by HPLC (200 Series,

Perkin Elmer, Shelton, CT, USA) on a Inertsil ODS-P 5µm, 4,6x150mm (GL Sciences Inc. Tokyo, Japan)

column by using water-acetonitrile binary gradient elution and time-programmed fluorescence detection.

In the majority of the samples naphthalene, acenaphthene and fluorene were not detected as a consequence of

160　their volatility and the elevated temperature of the sampling train. Therefore total PAH concentrations are given

in this study for 12 PAHs starting from 3-ring phenanthrene to 6-ring indeno(1,2,3-cd) pyrene. The extraction

efficiency was tested by sequential extraction of the same filter spots and it was found that >96 % of 3-6 ring

PAHs were extracted in the first round by applying the method described above. Recovery of PAHs from the

extract was also studied by performing the clean-up procedure on diluted PAH standard solutions. Average

165　recovery of 98 % (SD=8.9 %) was obtained for the 12 PAHs investigated in this study. The analytical results

were corrected for both extraction efficiency and recovery of the sample preparation.

3 Results and discussion

3.1 Emission factors for PM_{10} from waste burning

The PM_{10} EFs of the fuel wood and different solid wastes were found to be highly variable (from ~2.1 to ~81.5

170　mg g^{-1}), the lowest values being representative for the wood burning. In the literature, the $PM_{2.5}$ EF of wood

burning in traditional woodstoves varies between ~2.8 and ~13.3 mg g^{-1}, whereas the PM_{10} EF from a chimney-

type (eco-labelled) woodstove is smaller (~1.1−2.9 mg g^{-1}, Querol et al., 2016). Kistler et al., (2012) investigated

the PM_{10} EF of 12 Central European wood types from a chimney-type wood stove and found that it varied

between 0.3 and 3.6 mg g^{-1}. The values for turkey oak and black locust were 1.0 and 1.1 mg g^{-1}, respectively.

175　For the burning of oak logs in two different stoves Schmidl et al., (2011) reported PM_{10} EF of 1.2−1.4 mg g^{-1}. In

our measurements the PM_{10} EF from the burning of mixed turkey oak and black locust varied between 1.3 and

3.2 (on average 2.2 mg g^{-1}), indicating that the obtained EFs agree quite well with those reported in recent

studies.

Compared to the EFs of burning plastics, the PM_{10} EFs of burning wood-based materials were found to be

180　largely similar to each other, though the burning of OSB and LDF (both contain glue, the LDF surface coating

and edge tape as well) release twice as much PM_{10} (~3.2 mg g^{-1} for LDF, and 5.2 mg g^{-1} for OSB) into the

atmosphere as dry firewood under similar burning conditions (Fig. 3).

Figure 3 PM_{10} emission factors for burning of different waste types relative to the burning of dry fuel wood with a
185　**fixed experimental setup.**

The EF of burning rag is between the EF of PET and wood-based materials. This is not surprising as on average

46 % of the weight of the burned textile was cotton and 43 % was PET-based material. The EF for PET (on

average 11 mg g^{-1}) is roughly five times higher than that of fuel wood. The only available EF reported for PET

co-combustion in a boiler (though co-firing with beech log and in a different combustion unit of significantly

190　higher nominal power) has been 5.8 ± 0.5 mg g^{-1} (Tomsej et al., 2018), quite in agreement with our results. Our

findings indicate that the burning of a given mass of PE and/or PU release on average 1 order of magnitude more

PM_{10} into the atmosphere than wood burning. The combustion of PP, tyre and PVC is even more polluting as 16

times more PM_{10} is released into the atmosphere when burned under similar conditions. Somewhat surprisingly, our results also agree well with the few available EFs (for PM2.5) for the open air burning of mixed (presumably

195   mostly plastic) waste reported in the literature (10.5 and 7.37 mg g^{-1}) (Christian et al., 2010 and Jayarathne et al., 2018, respectively). The tyres already contain soot which might contribute to the increased PM_{10} emission of burning. The styrene containing materials yielded the highest EF among the investigated components. The PM_{10} EF for the burning of polystyrene was on average 53.1 mg g^{-1} and that of the expanded polystyrene was even higher (81.5 mg g^{-1}). On average the highest PM_{10} EF was obtained for the copolymer of styrene with butadiene

200   and acetonitrile (ABS). The absolute EFs for all waste types are summarised in Table 1.

Table 1 The absolute emission factors for PM_{10} from waste burning (in the unit of mg g^{-1}).

3.2 Emission factors for total PAHs from waste burning

The EFs of total PAHs (sum of 12 EPA priority pollutant PAHs from 3-ring phenanthrene to 6-ring

205   indeno(1,2,3-cd)pyrene) during the combustion of different waste types were determined (Table 2). Both wood and wood-based waste types were characterised by EFs below 5 mg kg^{-1}. The lowest EF was obtained for wood followed by paper, LDF and OSB. At least an order of magnitude higher EFs were measured for the other waste types investigated in this study. The average values covered a wide range from 15 mg kg^{-1} for rag to 257 mg kg^{-1} for ABS. The average total PAH EFs were similar for rag, PE, tyre and PET while it was higher for PVC,

210   PU and PP but remained under 100 mg kg^{-1} for all of these waste types. Tomsej and co-workers (2018) reported EFs for PE and PET co-combustion in a wood-fired boiler as being 4.5 ± 0.7 mg kg^{-1} and 10 ± 2.4 mg kg^{-1}, respectively, which agree relatively well with our results (22 and 32 mg kg^{-1}, respectively). The highest EFs were measured during the combustion of PS and ABS exceeding 100 mg kg^{-1}. The relatively high SD values are the consequence of varying conditions applied during the burning experiments as described in section 2.2.

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Table 2 Absolute emission factors of total PAHs for different waste types (mg PAHs kg^{-1} fuel).

In Figure 4 the EFs for total PAHs are shown for different waste types relative to wood burning. It is clearly visible that even the combustion of wood-based waste types (paper, LDF, OSB) generated considerably more (by a factor of 3–8) PAHs than the burning of wood. The relative PAH emissions increased by more than a factor of

220   50 in case of burning rag, PE, tyre and PET, and were well over 100 for PVC, PU, PP, PS and ABS. These extreme emission factors underline the severe hazard associated with the illegal burning of solid wastes in households.

Figure 4 Emission factors of total PAHs for different waste types relative to the burning of dry fuel wood with a fixed
225   **experimental setup.**

From the PM_{10} mass collected and the amount of total PAHs determined on the filter the PAH content of the PM_{10} fraction generated from each combustion experiment was calculated. The lowest mean PAH content of the PM_{10} aerosol was obtained for wood burning (0.16 µg total PAH in 1 mg PM_{10} aerosol) followed by the wood-based wastes OSB (0.52 µg mg^{-1}), paper (0.53 µg mg^{-1}) and LDF (0.53 µg mg^{-1}). For the other waste types the

230   mean PAH content increased from 0.70 µg mg^{-1} (tyre) through 1.0 µg mg^{-1} (PE), 1.4 µg mg^{-1} (PVC), 1.8 µg mg^{-1} (rag) to 2.2 µg mg^{-1} (PS). The highest PAH content was observed when burning PET (3.0 µg mg^{-1}), PU (3.0 µg mg^{-1}), ABS (3.1 µg mg^{-1}) and PP (3.2 µg mg^{-1}). It is clearly visible that the reactive share of PAHs in

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PM$_{10}$ emitted is typically an order of magnitude higher in plastic combustion as compared to wood burning. The relatively high PM$_{10}$ but low PAH emission factors of tyre combustion (as shown in Tables 1 and 2) can be explained by the presence of inorganic components in the material of tyres (e.g. inorganic fillers).

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It is important to note that the toxicity of PAHs varies from compound to compound. Among the 16 EPA priority pollutant PAHs five-ring compounds such as benzo(a)pyrene, dibenzo(a,h)anthracene, benzofluoranthenes, the 4-ring benzo(a)anthracene and the 6-ring indeno(1,2,3-cd)pyrene were found to be the most carcinogenic compounds. In order to compare the toxicity of different samples the toxic equivalency factors of PAHs are generally used. In this scale benzo(a)pyrene (BaP) has a value of 1 and the toxicity of the other 15 EPA priority pollutant PAHs varies from 0.001 to 1 (Nisbet and LaGoy, 1992). By applying these BaP toxicity equivalency factors the overall toxicity of the PAHs emitted during wood burning and the combustion of different wastes was compared (Table 3). The EFs of total PAHs as expressed in benzo(a)pyrene toxicity equivalent covered a very wide range of more than three orders of magnitude. The burning of 1 kg wood (oak) produced PAHs with total toxicity equivalent of 4.9 µg of BaP. The combustion of the same amount of wood-based wastes led to the emission of PAHs equivalent of about 30 times more BaP. The combustion of PE, rag, tyre, PET and PU resulted in the emission of PAHs 280−760 more toxic than the PAHs released from the burning of the same amount of wood. The toxicity equivalent EFs for PE and PET reported in Table 3 (1.4 and 2.4 mg kg^{-1}, respectively) show surprisingly good agreement with the only values available in the literature (1.1 ± 0.2 mg kg^{-1} and 1.8 ± 0.3 mg kg^{-1}, respectively) (Tomsej et al., 2018). Toxic PAHs in the highest amount were emitted from the combustion of PVC, PS, PP and ABS. The emission factors of total PAHs expressed in BaP toxicity equivalent were more than 3 orders of magnitude (!) higher for the combustion of these plastic wastes than the value obtained for wood burning. The very high emission of toxic PAHs from the combustion of plastic wastes as compared to wood burning may follow from the synergic combination of three factors:

1) On a per mass basis significantly more PM$_{10}$ is emitted from the combustion of plastic waste than from the burning of dry fuel wood (see Figure 3, Table 2);
2) PM$_{10}$ released from plastic waste combustion contains more PAHs per unit mass than PM$_{10}$ generated from wood burning;
3) The distributions of PAHs emitted from plastic waste burning differ markedly from that produced in wood burning. The combustion of plastic wastes results in the formation of the significantly more carcinogenic 4–6-ring compounds.

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Table 3 Comparison of the toxicity of PAHs emitted from wood burning and residential waste burning.

4 Conclusion

265 The PM$_{10}$ emission factors (EFs), total PAH emission factors and benzo(a)pyrene toxicity equivalent total PAH emission factors were determined for waste burning under controlled combustion conditions for the first time. The EF values were established for combustion of wood, paper, LDF, OSB, rag, PET, PE, PU, PP, tyre, PVC, PS and ABS which are all abundant waste types frequently burned in households and open fires in the developing countries and even in Europe. The determined PM$_{10}$ EF for wood burning was about 2.1 mg g^{-1} which shows

270 good correlation with the results of recent studies. The PM$_{10}$ EFs from the burning of other wood-based materials (paper, LDF and OSB) were only slightly higher, while from the combustions of rag and PET were

about five times higher than the EF of wood burning. The EFs for combustions of PE and PU were about 18 mg g^{-1} and 22 mg g^{-1} indicating nearly tenfold PM_{10} emission from burning of these waste types. Even higher PM_{10} EFs were determined during the combustion of PP, TR and PVC which varied between 33 mg g^{-1} and 35 mg g^{-1}.

275 Finally, the highest PM_{10} EF values were established for the combustion of PS and ABS which may reach 53 mg g^{-1} and 82 mg g^{-1}, respectively.

The total PAH emission factors varied in an even wider range. Burning of oak resulted in mean total PAH emission factor of 0.3 mg kg^{-1}, while wood based wastes (paper, LDF and OSB) produced mean total PAH emission factors from 1.1 to 2.7 mg kg^{-1}. Much higher total PAH emission factors were obtained for rag, tyre

280 and plastic wastes ranging from 15 to 257 mg kg^{-1}. These high total PAH emission factors are the consequence of the higher PM_{10} emission factors of plastic wastes as well as the higher total PAH content of unit mass of PM_{10} released during combustion. To compare the hazard of residential waste combustion with wood burning the total PAH EFs were converted into benzo(a)pyrene equivalent emission factors. It was found that the toxicity of PAHs emitted during the combustion of wood-based wastes (paper, LDF and OSB) was about 30 times higher

285 than that of PAHs released during the burning of oak. For rag, tyre and plastic wastes this ratio ranged between 280 and 4050 as a consequence of the different composition of PAHs emitted during the combustion of different waste types.

These results clearly show that residential waste burning (especially combustion of tyres and plastics) poses a serious hazard on human health for numerous reasons: waste combustion may produce considerably more PM_{10}

290 particles than wood burning, the particles contain more PAHs, and the PAHs formed during waste combustion are more carcinogenic than those released from wood burning. These findings underline the importance of concerted efforts of municipalities, authorities and NGOs to phase out the illegal burning of solid wastes in households. This would help significantly improve local air quality and reduce the number of limit exceedances of $PM_{10}/PM_{2.5}$. But more importantly, the health risks associated with air pollution would be reduced

295 disproportionately more than would follow from the reduction in $PM_{10}/PM_{2.5}$ concentrations due to the extremely hazardous composition of particulate matter emitted during the combustion of wastes in households.

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Figure 1 Solid waste specimens prepared for combustion tests.
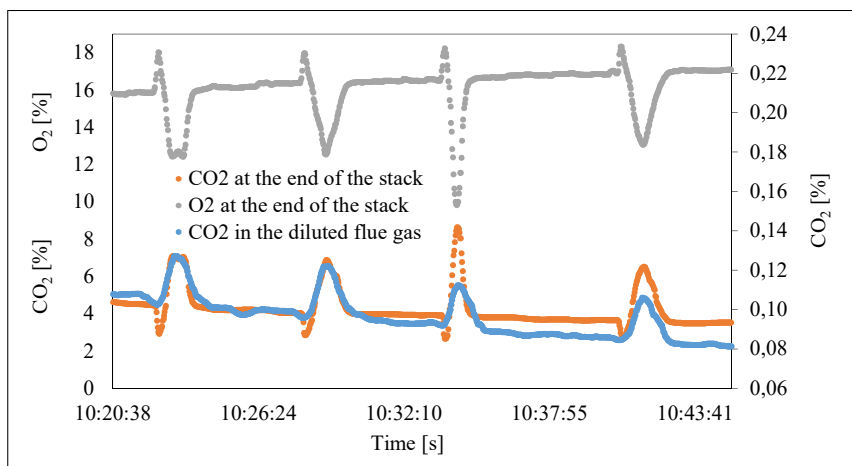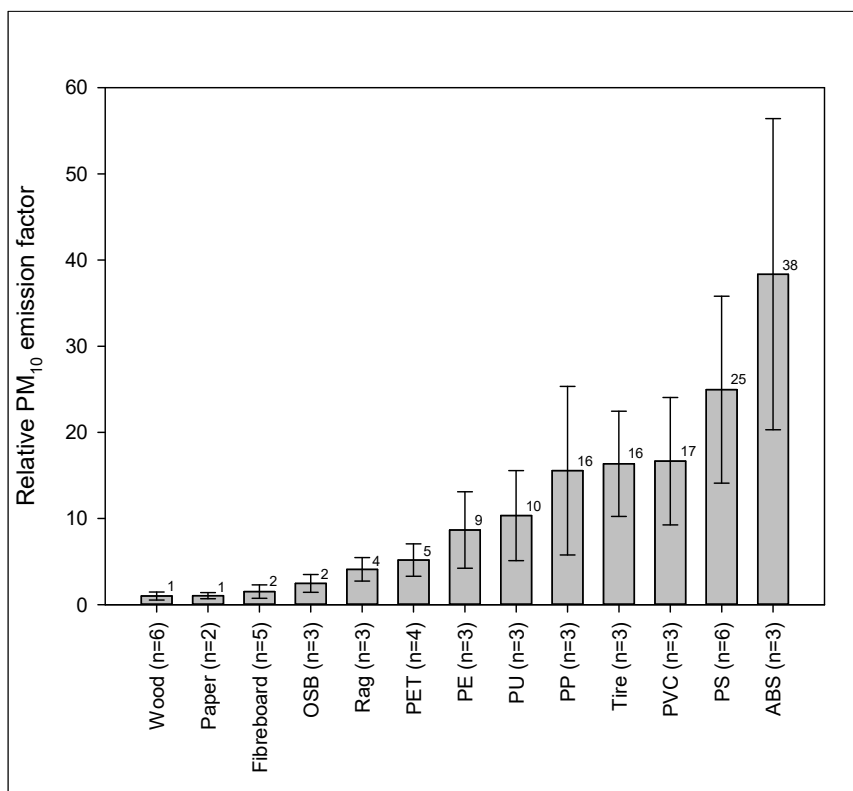
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Figure 2 Example (PS-F9 sample) for the variations of the measured CO$_2$ and O$_2$ mixing ratios at the end of the stack and the CO$_2$ mixing ratio in the diluted flue gas during combustions tests.

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Figure 3 PM$_{10}$ emission factors for burning of different waste types relative to the burning of dry fuel wood with a fixed experimental setup.

Table 1 The absolute emission factors for PM_{10} from waste burning (in the unit of mg g$^-$1).

Waste types	Mean	SD
WOOD	2.1	0.70
PAP	2.2	0.11
LDF	3.2	1.3
OSB	5.2	1.4
RAG	8.7	0.22
PET	11	1.6
PE	18	7.2
PU	22	8.4
PP	33	18
TR	35	6.0
PVC	35	10
PS	53	15
ABS	82	27

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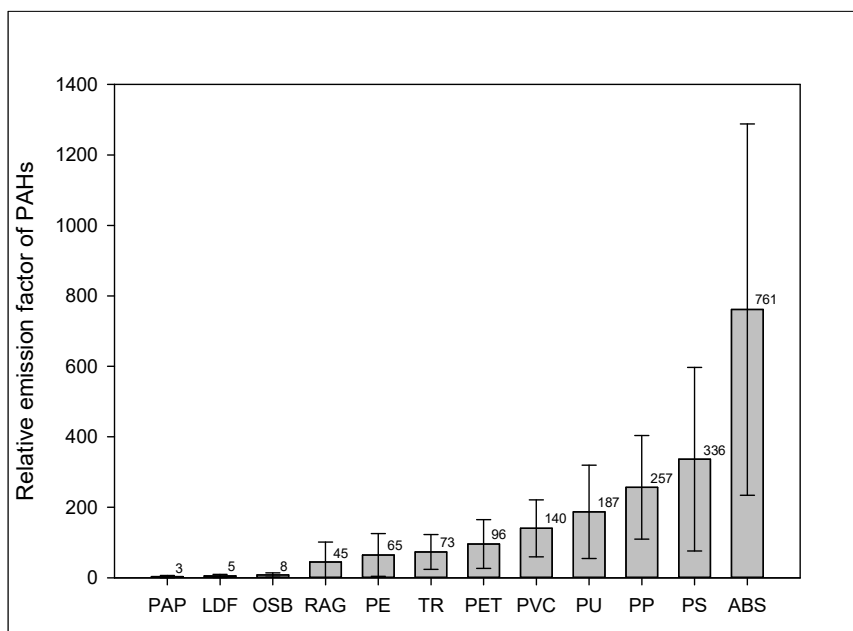
Table 2 Absolute emission factors of total PAHs for different waste types (mg PAHs kg^{-1} fuel).

Waste types	Mean	SD
WOOD	0.34	0.19
PAP	1.2	0.83
LDF	1.7	1.2
OSB	2.7	1.4
RAG	15	17
PE	22	16
TR	25	9.1
PET	32	14
PVC	47	5.1
PU	63	27
PP	87	8.3
PS	113	60
ABS	257	103

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Figure 4 Emission factors of total PAHs for different waste types relative to the burning of dry fuel wood with a fixed experimental setup.

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Table 3 Comparison of the toxicity of PAHs emitted from wood burning and residential waste burning.

Waste types	EF of Total PAHs expressed in BaP toxicity equivalent (mg kg^{-1} fuel)	SD (mg kg^{-1} fuel)	Relative to wood
WOOD	0.0049	0.0061	1
PAP	0.17	0.13	35
LDF	0.16	0.14	35
OSB	0.18	0.07	40
PE	1.4	1.2	280
RAG	2.1	2.1	420
TR	2.1	1.8	430
PET	2.4	1.4	490
PU	3.8	0.16	760
PVC	5.9	1.9	1200
PS	12	5.0	2300
PP	17	5.2	3400
ABS	20	12	4100

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