1 **Characterization of Gas and Particle Emissions from Laboratory**
2 **Burns of Peat**

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13
14 **Abstract**

15
16 Peat cores collected from two locations in eastern North Carolina (NC, USA) were
17 burned in a laboratory facility to characterize emissions during simulated field
18 combustion. Particle and gas samples were analyzed to quantify emission factors for
19 particulate matter (PM_{2.5}), organic carbon (OC), elemental carbon, light absorbing
20 carbon, absorption Angstrom exponent, metals, polycyclic aromatic hydrocarbons
21 (PAHs), and polychlorinated dibenzodioxins/dibenzofurans (PCDDs/PCDFs). CO
22 from the smoldering burns, up to 7 h in duration, contributed approximately 16% of
23 the total carbon emitted. Emission factors for black carbon (BC) and light absorbing
24 carbon (UVPM) were considerably lower than those found for forest litter burns.
25 Emission factors for PCDDs/PCDFs were near published values for forest fuels, at 1-4
26 ng toxic equivalents (TEQ)/kg carbon burned (Cb). Total PAH concentrations of ≥ 12
27 mg/kg were higher than published data from biomass burns, but roughly the same in
28 terms of toxicity. Application of these emission factors to the noteworthy 2008
29 “Evans Road” fire in NC indicates that PM_{2.5} and PCDD/PCDF emissions from this
30 fire may have been 4 -6% of the annual US inventory and 5% of the annual OC
31 amount.

32

33 **Keywords:** peat fires, PCDD/PCDF, PAHs, PM_{2.5}, CO, carbon

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35

36 **1 Introduction**

37 Peat, a major class of organic soil formed in wetlands, is a vast store of organic
38 carbon, widely distributed from polar to equatorial regions. Peat has high carbon
39 content and is particularly vulnerable to fire when drained (Turetsky et al., 2015).
40 Once ignited peat can burn undetected for very long periods of time (months to years),
41 propagating in a self-sustaining, low intensity, but widely distributed smoldering front
42 through the underground peat layers.

43 Emission factors (EFs) for above-ground fine fuels are well characterized compared to
44 the emissions from peat fires. The major gases produced during biomass burning
45 include CO₂, CO, methane (CH₄), volatile organic compounds (VOCs), and
46 particulate matter (PM). Emissions of CH₄, CO, and VOCs lead to the photochemical
47 production of ozone and secondary organic aerosols in the troposphere. Light
48 absorbing properties of carbonaceous PM as well as CO₂ and CH₄ are of interest for
49 their role as climate forcing agents. Smoldering combustion of surface fuels typically
50 shows distinctive emissions of PM and reduced trace gases (both higher and lower)
51 compared to flaming combustion of the same fuels (Bertschi et al., 2003),
52 underscoring the importance of characterizing smoldering peat combustion.

53 Some characterizations of emissions from peat fires have been performed, showing
54 higher PM_{2.5} and CO emissions per unit mass than for surface forest fuels (Geron and
55 Hays, 2013). In addition, since fuel consumption is so high in peat fires, PM_{2.5}, CO,
56 and CO₂ emissions are higher on an area basis compared to other fuel types. The
57 impact of PM_{2.5} emissions from peat fires is significant, as respiratory and cardiac
58 effects have been demonstrated in laboratory studies (Kim et al., 2014) and
59 epidemiological studies following exposure to peat wildfire smoke (Rappold et al.,
60 2011).

61 This study determines emission factors for underreported peat combustion pollutants
62 including fine particulate matter, light absorbing particles, polychlorinated
63 dibenzodioxins/dibenzofurans (PCDDs/PCDFs), and polycyclic aromatic
64 hydrocarbons (PAHs).

65

66 2 Materials and Methods

67 2.1 Peat collection and preparation

68 Peat samples were collected from each of two sites: Alligator River National Wildlife
69 Refuge (AR) and Pocosin Lakes National Wildlife Refuge (PL), both located in

Figure 1. Eastern North Carolina (U.S.A.) locations of peat core samples. Source: USGS, URL: http://nationalmap.gov/small_scale/atlasftp.html

70 Eastern North Carolina, USA (Figure 1). Each of these locations was the site of
71
72 extended wildfires in the last ten years that consumed over 16,000 ha each, burning
73 the organic soils to average depths of ~ 20 cm and emitting an estimated 0.44 Tg of
74 PM_{2.5} or 10% of annual PM_{2.5} emissions from all U.S. sources emission (Geron and

75 Hays, 2013). Further characteristics and description of the sites are available in
76 Supporting Information and elsewhere (Geron and Hays, 2013).
77
78 Peat core samples 150 mm in diameter and 200 - 250 mm deep were collected for
79 transport to an open burn test facility (OBTF) (Black et al., 2011; Gullett et al., 2008).
80 The cores were extracted from organic soil accumulated both near and 3 m from the
81 base of trees, where consumption was typically deepest during the AR and PL fires.
82 The cores were kept intact and stored vertically by transferring to a 150 mm diameter
83 galvanized stove pipe and capping with aluminium foil (see Fig. SI-1). All cores were
84 oven dried for 24-36 h at 110 °C to enable sustained combustion and to mimic the in
85 situ drying from proximal peat combustion. The cores were stored in plastic tubs for
86 approximately two months before burning. Multiple cores from each site were
87 homogenized before analysis for moisture, carbon, nitrogen, sulfur, chloride, PAH,
88 and PCDD/PCDF content (see Table SI-1).

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92 **2.2 Burn and sampling procedures**

93 Single peat cores (approximately 1.5 - 2.5 kg) contained within a galvanized stove
94 pipe were placed atop a pan balance (protected by a heat shield) surrounded by a hood
95 enclosure approximately 120 cm wide x 50 cm deep x 80 cm high to concentrate the
96 smoke for sampling (see Figure SI-2). The hood enclosure was located in the 3.9 x 3.9
97 x 4.6 m ventilated OBTF, described more fully elsewhere (Black et al., 2011). An
98 induced draft fan refreshed the OBTF volume every 90 seconds, thereby simulating
99 open combustion conditions. The peat cores were ignited briefly (~30 sec) in the
100 center of the upper surface with a propane gas torch.

101 Sampling of the flaming/smoking peat from the hood enclosure commenced after one
102 minute, using continuous emission monitors (CEMs, see Table SI-2) for CO₂, CO,
103 oxygen (O₂), and total hydrocarbon (THC) as CH₄, as described more fully elsewhere
104 (Woodall et al., 2012). All CEMs were zeroed and spanned at the beginning of each
105 day and rechecked at day's end. For PCDD/PCDF and PAH determinations, smoke
106 was drawn from the outlet of this hood enclosure at a flow rate of approximately 0.6
107 m³ min⁻¹ through a 25 x 20 cm quartz filter (see Fig. SI-3), a cylindrical pre-cleaned

polyurethane foam (PUF) plug (7.5 cm diameter), and XAD-2 resin in series, consistent with U.S. EPA sampling method TO-9A (EPA, 1999). Collection of at least 10-15 g of carbon as CO₂ was targeted to minimize non-detects among the 17 Cl-substituted PCDD/ PCDF congeners in the toxic equivalency (TEQ) measure.

PM_{2.5} was sampled from the enclosure hood using impactors with both polytetrafluoroethylene (PTFE, pore size of 2.0 µm) and quartz filters. Both samplers used a Leland Legacy sample pump (SKC Inc., Pittsburgh, PA, USA) with a constant airflow of 10 L/min. The Leland Legacy sample pumps were calibrated with a Gillibrator Air Flow Calibration System (Sensidyne LP, St. Petersburg, FL, USA) as the primary standard. An internal flow sensor on the Leland pump measured flow directly and acted as a secondary standard to maintain the set flow constant. Four sampling inlets for the CEMS were located above the PM_{2.5} and OC/EC impactors (see Fig. SI-2).

Fires were typically self-sustaining; an initial 1-3 min flaming phase after propane torch ignition was followed by prolonged smoldering. In two cases, the peat had to be relit. Burns were comprised of an initial and final period of sampling, each period approximately 3-4 h in duration and designated as “0-3h” and “3-7h”. Sampling was typically 3 hours followed by sorbent and filter changes and then an additional 4 to 4.5 hours sampling, after which smoldering was still evident.

2.3 Sample analysis

2.3.1 PM, metals, and organic carbon/elemental carbon

PM_{2.5} samples were collected on quartz and pre-weighed PTFE filters in PM_{2.5} impactors. Gravimetric determination of PM was by microbalance (resolution 0.01 mg). Metal analysis on PM_{2.5} PTFE filters was carried out using a wavelength-dispersive X-ray fluorescence spectrophotometer (PANalytical PW2404, Westborough, MA). Thermal-Optical Analysis (TOA) using a modified NIOSH 5040 method was performed to determine the organic carbon (OC), elemental carbon (EC), and total carbon (TC) content in the PM collected (Khan et al., 2012).

2.3.2 BC and UVPM

Black carbon (BC) and light absorbing organic compounds were sampled using a portable, filter ticket-based, continuous, optical aethalometer technique (Models AE51 and AE52, Aethlabs, San Francisco, CA, USA) based on light attenuation at 880 nm (BC) and 370 nm (UVPM). Enhanced aerosol absorption in the UV portion of the spectrum for biomass burning emissions is indicative of the presence of light absorbing organic carbon, i.e. brown carbon. Brown carbon emissions are quantified by the AE-52 as UVPM, which is defined as the BC mass equivalent excess attenuation at 370 nm over that ascribed to BC. For comparison with other measurement methods (Ajtai et al., 2011), optical attenuation attributed to brown carbon is also reported as an absorption angstrom exponent, AAE:

$$AAE = - \ln [(880/370) * (1 + UVPM/BC)] / \ln (370/880)$$

2.3.3 PCDD/PCDF and PAH

PCDD/PCDF samples from the raw peat and emissions were cleaned up and analyzed using an isotope dilution method based on U.S. EPA Method 23 (EPA, 1996). Concentrations were determined using high resolution gas chromatography/high resolution mass spectrometry (HRGC/HRMS) with a Hewlett-Packard gas chromatograph 6890 Series coupled to a Micromass Premier mass spectrometer (Waters Corp., Milford, MA, USA) with an RTX-Dioxin 2, 60 m × 0.25 mm × 0.25µm film thickness column (Restek Corp., Bellefonte, PA, USA).

PAHs were cleaned up according to U.S. EPA Method TO-13 (1999) and quantified using an internal standard method based on Method 8270 of SW-846 (EPA, 2007). Samples were analyzed using a Hewlett-Packard gas chromatograph 5890 Series II with a DB-5 60 m × 0.25 mm × 0.25µm column (Agilent/J&W Scientific) connected to a Hewlett Packard 5972 mass spectrometer. PAH emission factors were evaluated using toxic equivalent factors (TEFs) relative to the benzo(a)pyrene toxicity equivalent (B(a)P-TEQ) (Larsen, 1998).

2.3.4 Emission Factors

Emission factors were calculated using the carbon mass balance approach (Ward et al., 1979) as is common for sources of open burning (Black et al., 2011). This method calculates the target compound mass in ratio to the carbon mass burned from concurrently collected CO, CO₂, and CH₄ measurements. PM carbon was neglected as it amounted, on average, to less than 0.1% of the carbon emitted. These values of pollutant mass per mass of carbon burned, along with the carbon percentage in the raw peat (see Table 1), were used to derive emission factors in terms of mass of target compound per dry weight mass of peat biomass burned in g/kg.

3 Results and Discussion

Four 5 to 7 h (typical) burns were conducted on 2.1 kg (SD = 0.6 kg) peat samples from each of the two sites. The total gravimetric mass loss averaged 0.176 kg (~ 10% of the initial weight) for the AR peat and about 0.163 kg (~ 7%) for the PL. Analyses of six peat samples from each source indicated similar carbon contents but considerable variability: 28% (10% s.d.) for AR and 30% (8% s.d.) for PL. These peat samples had distinctively lower C content than previous cores from this same general area (Geron and Hays, 2013), reflected in higher ash values: 44% (10% s.d.) for AR and 62% (12% s.d.) for PL. Assuming half of this mass loss was attributed to loss of carbon from biomass, then 19% of the AR carbon and 11% of the PL carbon, respectively, would have been released during the burn, indicating an upper limit on the extent of the carbon burned. The average mass loss rate over the course of the burn for AR and PL was 0.92 g min⁻¹ and 0.52 g min⁻¹, respectively.

The run-averaged modified combustion efficiency, MCE ($\Delta\text{CO}_2/(\Delta\text{CO}_2 + \Delta\text{CO})$) (Ward and Radke, 1993) ranged between 0.80 and 0.88 for the individual AR and PL samples, with no notable distinction between the two peat sources for the 0-3 h and 3-7 h samples. These MCE values are much lower than the values found for over 50 laboratory combustion tests of field-gathered biomass (none were peat) averaging over 0.94 (Hosseini et al., 2013), but are consistent with lower MCE values for peat burns found elsewhere (Christian et al., 2003; Liu et al., 2014; May et al., 2014; Rein et al., 2009; Stockwell et al., 2014; Yokelson et al., 1997).

199 The temporal combustion characteristics for the peat burns were distinctive from
 200 those observed in the same burn facility with mixed pine understory biomass (Aurell
 201 and Gullett, 2013). Figure 2 shows the evolution of ΔCO , ΔCO_2 , and MCE
 202 throughout the beginning of an illustrative PL peat burn in comparison with the pine
 203 burn. The ΔCO_2 and ΔCO concentrations for the PL peat burn decreased steadily
 204 after a brief (< 60 s) flaming period while the MCE remained relatively constant
 205 throughout. In contrast, the forest litter showed declining MCE and gases throughout
 206 its burn, reflective of a gradual transition from flaming to smoldering.

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209

210 Figure 2. Comparison of concentration and modified combustion efficiency profiles
 211 for peat burns (-p, solid lines) (this work) and forest litter burns (-f, dashed
 212 lines)(Aurell and Gullett, 2013).

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214

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216 **3.1 Summary Emission Factors**

217

218 Table 1 includes the average emission factors and standard deviations from the burn
219 tests. A breakdown of these results follows in the sections below.

220

221 Table 1. Summary Emission Factors for Peat Burns from Alligator River (AR) and Pocosin Lakes (PL) National Wildlife Refuges.

		AR				PL			
		Avg		Avg	sd	Avg		Avg	sd
Units		0-3 h	3-7 h	0-7 h		0-3 h	3-7 h	0-7 h	
PM _{2.5}	g/kg	10.4	3.7	7.1	5.6	10.8	1.0	5.9	6.7
	g/kg Cb	36.8	13.2	25	20	36.5	3.4	20	22.6
CO	g/kg	111	104	108	20	129	116	122	14
CO/C	g/g	0.39	0.37	0.38	0.07	0.44	0.39	0.41	0.05
CO ₂	g/kg	865	856	861	112	941	902	922	47
MCE	Unitless	0.86	0.86	0.86	0.02	0.82	0.83	0.83	0.02
BC	g/kg	0.08	0.04	0.07	0.04	0.06	0.06	0.06	0.01
UVPM	g/kg	0.19	0.11	0.15	0.11	0.16	0.19	0.18	0.05
BC/UVPM	Unitless	0.43	0.38	0.45		0.35	0.29	0.32	
OC	g/kg	8.7	3.3	6.3	4.1	5.6	2.1	4.3	4.1
EC	g/kg	0.088	0.017	0.052	0.057	0.103	0.061	0.082	0.091
OC/EC	Unitless	99	194	121		54	143	52	
BC/EC	Unitless	14	3.3	1.9		0.63	2.1	0.97	
AAE	Unitless	2.03	2.28	2.15	0.61	2.13	2.47	2.30	0.44
THC as CH ₄	g/kg	31.0	24.1	30.2	7.2	31.8	33.1	32.4	14.2
PAHs (16)	mg/kg	70	8.1	16	8.9	22	6.1	14	9.0
	mg TEQ/kg, B[a]P eq.	0.82	0.04	0.16	0.05	0.20	0.09	0.15	0.07
PCDD/PCDF	ng/kg	310	240	510 ¹	280	970	160	560	490
	ng TEQ/kg	1.9	2.2	2.6 ¹	1.2	3.1	0.6	1.9	1.6
	ng TEQ/kg Cb	2.5	2.9	3.4	1.6	2.9	0.6	1.8	1.4

222

223 ¹The 0-7 h average is higher than the 0-3 h and 3-7 h averages because two more single, undivided 0-7 h samples were included.

224

225 **3.2 PM_{2.5}**

226 Minimal difference was observed between the PM_{2.5} emission factors from each peat
227 type (see Table 1), given the variability in burn results. The average 0-7h PM_{2.5}
228 emission factor for both peats is 6.6 ± 5.9 g PM_{2.5}/kg biomass burned. Literature
229 values for peat PM_{2.5} emission factors are all higher than observed in this work with
230 values of 33 and 44 g/kg biomass for German peat (Iinuma et al., 2007) and 35 g/kg
231 for PM₁ from Indonesian peat (May et al., 2014). Previous sampling of ground fires
232 at AR and PL (Geron and Hays, 2013) resulted in PM_{2.5} emission factors of 34-66
233 g/kg. These higher emissions may partly be reflective of the large distinctions in
234 carbon content (54-60% in their work and 28-30% in our work) and ash content (13-
235 17% in their work and 44-62% in our work). This is likely due to our samples being
236 gathered from areas that experienced previous fires, suggesting significant differences
237 in the peat sources despite collection from the same burned areas. Another potentially
238 important difference with others' results is that our sampling times of up to 7 h were
239 much longer than the sampling times of 57 min reported by others (Iinuma et al.,
240 2007). The average PM_{2.5} emission factors in this work were three to ten times higher
241 for the initial three hour sampling period than for the final period for both peat types
242 (Table 1). To look at the early burn performance, we examined two samples from the
243 top 2 cm of each ~20 cm core type for carbon content and found no distinction with
244 the whole sample values that could explain this PM_{2.5} variation. This suggests that the
245 emissions at longer sampling times may be a product of reaction regimes that were
246 controlled by different mechanisms (e.g., diffusion) than at the burn onset (Bertschi
247 et al., 2003).

248

249 **3.3 Inorganics in PM**

250 Metal concentrations on 14 PM_{2.5} PTFE filters were determined by XRF (see Table
251 SI-3). The most abundant elements in the filter-collected PM mass aside from carbon
252 and oxygen were sulfur and chlorine, where the sulfur fraction averaged over four
253 times higher for AR peat (1.96%) than for PL peat (0.46%). Some filter elements
254 commonly found in soil (e.g., Si, Fe, Ca, Sr, Cu, Cr, and Al) were observed after
255 background corrections. The resulting Cl emission factors are 0.011 and 0.004 g Cl/kg
256 biomass burned (0.051 and 0.012 g Cl kg⁻¹ carbon burned (C_b)) for AR and PL,

257 respectively. These values, when compared against the raw peat Cl analyses, are
258 consistent with data from a variety of fuels (Christian et al., 2003; Hosseini et al.,
259 2013). Peat from AR had a higher chlorine concentration than peat from PL, at 460
260 and 190 mg/kg, respectively, which may be due to its proximity to the Atlantic Ocean
261 and subsequent salt water intrusion during storm events. However, the top 2 cm of
262 the peat cores had lower Cl content, at 180 and 70 mg/kg for AR and PL, respectively.
263 Sulfur was detectable at AR (0.12 mg/kg) but was below the detection limit at PL
264 (<0.05 mg/kg).

265

266 **3.4 OC, EC, BC, and UVPM**

267

268 The overall AR OC emission factor, 6.3 ± 4.1 g/kg, was indistinguishable from the
269 overall emission factor of PL, 4.3 ± 4.1 g/kg. For both peat sources, the initial 0-3h
270 samples had approximately two to three times higher OC emission factors than the
271 final 3-7h samples, similar to observed changes in PM_{2.5} mass emission factors. EC
272 emission factors for AR and PL were 0.05 g/kg and 0.08 g/kg, respectively, much
273 lower than reported for biomass burns from others (Hays et al., 2002). Our averaged
274 OC/EC ratios were 115 and 87 for AR and PL, respectively, a bit lower than the value
275 of 151 for Indonesian peat (Christian et al., 2003). However, the OC/EC ratios from
276 this work were substantially higher than the OC/EC ratios in the range of 1.4 to 14
277 from more recent peat burning studies in the laboratory (Iinuma et al., 2007; See et al.,
278 2007). This wide variation in reported OC/EC ratios may be influenced by different
279 sample collection procedures, where values are expected to change with downwind
280 travel due to continued plume reactions. Associations of OC emission factors with
281 MCE were previously found to be negative with laboratory burns of multiple biomass
282 species (Hosseini et al., 2013). EC correlations with MCE were minimal (Hosseini et
283 al., 2013) or weakly positive (McMeeking et al., 2009), while prescribed biomass
284 burns have found positive correlations with light absorbing species (Aurell and
285 Gullett, 2013).

286

287 Average BC emission factors
288 for the AR and PL sources
289 (Table 1) were essentially
290 equivalent at 0.07 g/kg (0.25 g
291 kg⁻¹ C_b and 0.06 g/kg (0.20 g
292 kg⁻¹ C_b) respectively, while
293 the emission factors for UVPM
294 show some separation: 0.15
295 g/kg versus 0.18 g/kg. The
296 value for BC can be compared
297 with the values obtained by

298 others (May et al., 2014), who
299 obtained 0.01 g/kg for

300 Indonesian peat. The much

301 larger BC EF measured here compared to May et al. (May et al., 2014) may be partly
302 due to the different BC measurement approach used in each study, which can result in
303 a difference of a factor of 2 (Yelverton et al., 2014). A typical AAE curve is shown in
304 Figure 3, indicating the decline in attenuation attributed to brown carbon with
305 continued smoldering. AAE values were slightly higher for PL than for AR with
306 overlap in the one standard deviation range. These AAE values are somewhat lower
307 than the values that have previously been reported for laboratory combustion of peat,
308 3 – 6 (Liu et al., 2014). However, given the wider wavelength range (370 – 880 vs.
309 405 – 781) used to calculate AAE and the tendency for AAE to decrease when
310 including wavelengths in the IR, the lower values are reasonable. These values are
311 supported by results of Chen et al. (Chen et al., 2010) who observed low AAE (~1.5)
312 for this wavelength range applied to smoldering combustion of moist fuels.

313 Additionally, AAE tends to increase with decreasing MCE, therefore the higher AAE
314 measured by Liu et al. are consistent with the lower MCEs they observed for peat.

315 BC and UVPM values for AR were higher during period 0-3h than 3-7h, while little
316 distinction was noted for the PL. For both peat sources, the UVPM/BC ratio is about
317 2-3:1.

318

Figure 3. Angstrom Absorption Exponent through the initial period of a representative peat burn, PL, 0-3 h.

319 Whole-burn BC and UVPM values from field forest burns and laboratory burns are
 320 typically greater than 0.5 g/kg (approximately 1 g kg⁻¹ C_b) (Aurell and Gullett, 2013),
 321 higher than the peat results observed here (Table 1). Comparison of BC/UVPM ratios
 322 for peat and forest litter burn profiles with illustrative data (Figure 4, logarithmic
 323 scale) shows that UV absorption in forest fires is dominated by the BC whereas in the
 324 peat system, absorption is dominated by UVPM. This observation is consistent with
 325 greater BC production from the flame-dominant forest litter burn and the larger OC
 326 content of emissions from the peat smoldering.

327
 328 Figure 4. Comparison of BC/UVPM ratios in peat (this work, representative data) and
 329 forest litter burns ((Aurell and Gullett, 2013))

330
 331
 332 Overall, BC and UVPM emission factors and MCEs for peat are significantly lower
 333 than for the forest biomass and are consistent with others' findings (McMeeking et al.,
 334 2009). Both the BC and UVPM values appear independent of MCE, at the lower
 335 MCE range observed for peat combustion (Figure 5). However, laboratory burns
 336 (Aurell and Gullett, 2013) with forest biomass that allow continuous determination of
 337 MCE values rather than whole-run composite MCE determinations show BC and
 338 UVPM emission factors of 1.0 and 0.4 g kg⁻¹ C_b, respectively, when the burn profiles

are at an MCE of 0.85 (Figure 3C (Aurell and Gullett, 2013)), suggesting that the lower BC and UVPM values observed in these tests are typical for smoldering conditions, regardless of the biomass type. The distinctions in peat emissions versus pine litter burns are underscored by a visual comparison of the PM_{2.5} mass collected on the filters. The peat filters are a deep yellow (Fig. SI-3), reflecting the relatively high organic to black carbon content while conversely, the filters from pine burns are black to gray, depending on the combustion efficiency.

Figure 5. Run-averaged BC and UVPM emission factors compared to combustion quality (MCE) from peat burns (diamonds, this work) and forest litter burns (circles, Aurell & Gullett, 2013).

3.5 PAHs

PAH emissions (Table 2) from combined gas and particle samples were dominated by naphthalene, fluorene, and phenanthrene, with all three compounds exceeding 9 mg/kg C_b for both AR and PL. A more extensive analysis of particle phase PAHs beyond the 16 “EPA PAHs,” is reported elsewhere (George, 2015). The five most prevalent PAHs are also found in the top eight PAHs for pine combustion measurements (Jenkins et al., 1996). The total PAH emissions (sum of 16 compounds)

360 of 16.0 ± 8.9 mg/kg for AR and 14 ± 9.0 mg/kg for PL are similar to each other and
361 consistent with values (18 and 25 mg/kg) for peat from (Iinuma et al., 2007).

362 There is relative similarity in the PAH compositional profile and emission factors for
363 the two sites, although the PAH total of PL was twice the PAH total of AR (Table 2).
364 The burn emissions were over 100 times greater than the concentrations in the raw
365 peat analyses (on a C basis), confirming that the emissions were not solely due to
366 vaporization but were primarily formed during combustion. The initial period of the
367 burns had four to nine times higher PAH emission factors compared to the final
368 period of the burns for both peat types, which was consistent with observed trends in
369 $PM_{2.5}$ and OC emission factors.

370 The toxic equivalency factors were applied to the 12 carcinogenic PAHs to derive an
371 overall hazard level. Our B[a]P equ. toxicity values, 0.12 ± 0.04 mg/kg and 0.16 ± 0.08
372 mg/kg for AR and PL, respectively, are above values reported for Indonesian and
373 German peat (~ 0.06 mg B[a]P equ. kg^{-1} for both sources) (Iinuma et al., 2007).
374 However, our values are more consistent with values calculated from other biomass
375 burns in the range of 0.1 to 0.57 B[a]P equ./kg (Dhammapala et al., 2007; Jenkins et
376 al., 1996). The initial 0-3h emissions were 2 to 20 times more carcinogenic (Table 2)
377 than the emissions in the 3-7h samples, and the AR emissions were more than three
378 times as carcinogenic as those from PL.

379

380 Table 2. PAH Emission Factors from Raw Peat and Burning Peat.

	Raw peat biomass mg kg ⁻¹		PAH Concentrations mg kg ⁻¹ carbon. N=3												Relative Potency*		
			ARNWR				PLNWR										
			0-3 h		3-7 h		Total 0-7		0-3 h		3-7 h		Total 0-7				
			ARNWR	PLNWR	mean	SD	mean	SD	h	s.d.	mean	SD	mean	SD		h	s.d.
Naphthalene	0.0287	0.024	119.19	19.77	18.17	6.99	28.95	13.18	25.88	7.07	6.86	7.07	16.37	12.19	0.000		
Acenaphthylene	N.D.	0.0013	26.96	6.04	0.94	0.44	4.96	4.34	2.33	1.26	0.35	0.33	1.34	1.36	0.000		
Acenaphthene(CCC)	N.D.	ND	7.20	0.52	0.69	0.24	1.55	0.74	1.29	0.14	0.66	1.00	0.98	0.72	0.000		
Fluorene	N.D.	N.D.	61.99	5.36	5.58	2.59	13.12	6.74	13.77	1.45	4.24	6.13	9.00	6.56	0.000		
Phenanthrene	0.0203	0.0139	63.74	2.30	8.74	3.28	14.99	5.42	13.70	1.48	3.50	4.53	8.60	6.35	0.001		
Anthracene	0.0017	0.0025	9.61	0.15	0.72	0.29	1.96	1.02	2.63	0.82	0.64	0.90	1.64	1.34	0.001		
Fluoranthene(CCC)	0.0068	0.0076	11.73	1.26	1.09	0.55	2.50	1.31	2.08	0.10	0.50	0.56	1.29	0.93	0.050		
Pyrene	0.0051	0.0088	13.33	0.93	1.23	0.49	2.84	1.40	2.28	0.27	0.81	1.10	1.55	1.08	0.001		
Benzo(a)anthracene	0.0017	0.0076	3.32	0.17	0.22	0.10	0.66	0.37	0.67	0.20	0.31	0.47	0.49	0.38	0.005		
Chrysene	0.0085	0.0164	6.18	0.15	0.69	0.24	1.38	0.57	1.49	0.20	0.42	0.61	0.96	0.71	0.030		
Benzo(b)fluoranthene	N.D.	0.0252	1.57	0.15	0.09	0.03	0.31	0.19	0.33	0.09	0.10	0.13	0.21	0.16	0.100		
Benzo(k)fluoranthene	N.D.	0.0139	1.00	0.13	0.06	0.03	0.23	0.13	0.18	0.02	0.05	0.07	0.11	0.09	0.050		
Benzo(a)pyrene(CCC)	0.0034	0.0088	1.72	0.17	0.07	0.03	0.32	0.22	0.27	0.04	0.15	0.22	0.21	0.16	1.000		
Indeno(1,2,3-cd)pyrene	0.0034	0.0101	0.90	0.17	0.04	0.01	0.17	0.13	0.11	0.03	0.06	0.07	0.09	0.05	1.100		
Dibenz(a,h)anthracene	N.D.	0.0013	0.10	0.00	0.02	0.00	0.04	0.01	0.04	0.01	0.04	0.00	0.04	0.01	0.020		
Benzo(ghi)perylene	0.0017	0.0063	0.91	0.15	0.05	0.01	0.18	0.12	0.13	0.05	0.07	0.08	0.11	0.06	0.100		
Total	0.081	0.148	329		38		74.1		67		19		42.9				
B[a]P eqs., mg/kg#	0.008	0.025															
B[a]P eqs, mg/kg Cb			2.9		0.16		0.56		0.67		0.30		0.49				
B[a]P eqs, mg/kg			3.84		0.21		0.75		0.62		0.28		0.45				

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382

383 N.D. = Non-detect; PAHs in Bold are classified as reasonably anticipated human carcinogens (NTP, 2014)

384 * Larsen, 1998

385 #Raw peat mass, dried

3.6 PCDDs/PCDFs

The PCDD/PCDF emission factors for AR and PL, respectively, were 2.6 ng TEQ/kg and 1.9 ng TEQ/kg, as shown in Table 1. These similar emission factors exist despite a higher chloride concentration (203 mg kg⁻¹ versus 112 mg kg⁻¹, Table SI-1) in the raw AR biomass. These values are apparently contrary to observations previously noted for biomass combustion (Grandesso et al., 2011) where four-fold changes in Cl content affected PCDD/PCDF yields. The PCDD/PCDF emissions easily exceed the respective raw peat content, confirming that PCDDs/PCDFs are formed during peat combustion rather than simply volatilized (Black et al., 2012b; Black et al., 2012c).

The overall carbon-weighted emission factor of 10 ng TEQ kg⁻¹ C_b (not shown) is in the upper range of values observed for six prescribed and laboratory burns of pine forest understory biomass in the range 0.2 to 9.2 ng TEQ kg⁻¹ C_b (Aurell and Gullett, 2013). This indicates that the temperatures favouring PCDD/PCDF formation (300-400 °C) during smoldering peat burns favor PCDD/PCDF formation compared to flaming forest litter burns.

The peat PCDD/PCDF emission factor from the laboratory burns is noticeably higher than most field-measured forest litter values (Aurell and Gullett, 2013; Black et al., 2011). Indeed, the peat emission factor is significantly higher than fourteen other emission factors ranging from 0.5 to 2.5 ng TEQ kg⁻¹ C_b from varying biomass types (grass understory, pine forest litter, deciduous forest litter, deciduous forest) and in both field and laboratory sampling (Black et al., 2011; Black et al., 2012a).

The initial 0-3h PL samples had approximately five times higher PCDD/PCDF emission factor than the 3-7h samples (Table 1), consistent with the higher initial PM_{2.5}, OC, and PAH emission factors (Table 1) in this work. However, this distinction was not as noticeable for the five AR samples. Higher initial PCDD/PCDF values contrast with data from laboratory forest biomass burns in which the post-flame smoldering stage yielded 10-fold higher emissions per unit of carbon combusted (Gullett et al., 2008). A shift in the chemical mechanism of formation seems an unlikely explanation for these results as the duration of the burn is dominated by a constant smoldering phase. Comparison of the peat mass loss rate with that of the carbon collected by the constant volume samplers appears linear (not shown), suggesting little change in the mechanism of smoldering throughout the

418 experiments. A consistent mechanism of formation supports observations of the
419 2,3,7,8-Cl-substituted congener patterns, which are relatively constant whether taken
420 in the initial or final portion of the burn (Fig. SI-4) for both AR and PL. In all
421 samples, the pattern is dominated by the OCDD and then the HpCDD congeners (Fig.
422 SI-5); when the TEF-weighted values are compared, the 1,2,3,7-8-PeCDD, 2,3,7,8-
423 TeCDD, and 2,3,4,7,8-PeCDF congeners yield, in order from highest to lowest, the
424 highest contribution to the TEQ toxicity measure (Fig. SI-6).

425 **3.7 Perspective**

426
427 A peat fire at PL NWR during the spring and summer of 2008 (the “Evans Road”
428 wildfire) impacted 16,813 ha and consumed carbon at a rate of 545 tons C/ha (Mickler
429 et al., 2011) with on-site estimates ranging from 200-600 tons C/ha. This carbon/area
430 consumption is higher than values reported for Scottish Highlands peat fires (96 ± 15 t
431 C/ha, (Davies et al., 2013)) and western Canada peat fires (21 t C/ha, (Benscoter and
432 Wieder, 2003) and 32 t C/ha, (Turetsky and Wieder, 2001)). An earlier fire in the PL
433 NWR (1985) estimated peat burn depths up to 0.1 m, resulting in a carbon/area
434 estimate up to 119 t C/ha (Poulter et al., 2006). This fire, however, was of
435 significantly shorter duration (approximately 14 days) than the 2008 fire
436 (approximately 3 months) and likely resulted in shallower burn depths. Carbon
437 consumption is determined by the water table height and the peat composition, the
438 latter which is particularly heterogeneous on these mixed peat and mineral soils
439 ((Poulter et al., 2006)), and which offers an explanation for these widely disparate
440 carbon/area values. This range of consumption rates, when combined with the low 0-
441 7 h emission factors from each site, enabled a prediction of the emissions from the
442 Evans Road fire. Even when using the lower value of the range of carbon/area
443 estimate (200 t C/ha), comparison of these emission values with US EPA source
444 inventories suggests that this single peat fire contributed 4 to 6 % of the annual $PM_{2.5}$
445 (EPA, 2015) and PCDD/PCDF (EPA, 2013) emissions and about 5% of the annual
446 estimated OC emissions (EPA, 2012).

449 **4 Conclusions**

450 Smoldering peat combustion was characterized by comparatively low MCE values
451 (0.80-0.88) with CO and CO₂ levels that were generally constant throughout the
452 multi-hour burns. This pattern is distinct from combustion of other biomass where the
453 MCE, CO, and CO₂ peak rapidly during the flaming phase initially, only to tail off
454 quickly after combustion of volatiles and the onset of fixed carbon combustion and
455 smoldering. Despite these seemingly static conditions, emission factors for PM_{2.5},
456 OC, PAHs, and PCDDs/PCDFs showed significant reductions in the second half of
457 the peat burns.

458 PM_{2.5}, BC, and UVPM emission factors in this work were much lower compared to
459 previous laboratory- and field-based biomass burning studies, perhaps due to the low
460 C content of our peat. Little distinction was noted between the PM_{2.5} emission factors
461 for the two peat sources.

462 The trend of increasing BC and UVPM with increasing MCE noted during forest
463 biomass burns was not apparent with these smoldering-dominated peat burns, as their
464 values in this work were lower and covered only a limited MCE range, below that
465 associated with flaming combustion (>0.90). The OC/EC ratios observed for this
466 work were much higher than for forest biomass burns, where higher EC values are
467 observed during prolonged flaming combustion.

468 PAH emissions were highest during the first half of the burn, resulting in emission
469 factors higher than published biomass burn results. In toxicity equivalents, emission
470 factors were consistent with other published peat and biomass data. PCDD/PCDF
471 emission factors were on the high end of the range of published values for biomass
472 combustion.

473 Calculations for a single peat fire in North Carolina using the emission factors
474 determined here showed that this single fire may have accounted for 3 to 6% of the
475 annual PM_{2.5} and PCDD/PCDF emissions in the U.S. and 5% of the OC emissions.
476 While peat fires are rare compared to large forest and rangeland fires, their range of
477 emission factors, burn areas, and carbon consumption data indicate that they can be a
478 major source of air pollution suggesting additional studies to improve the certainty
479 of emission estimates.

480

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502 **References**

503 Ajtai, T., Filep, A., Utry, N., Schnaiter, M., Linke, C., Bozoki, Z., Szabo, G., and
504 Leisner, T. 2011. Inter-comparison of optical absorption coefficients of atmospheric
505 aerosols determined by a multi-wavelength photoacoustic spectrometer and an
506 Aethalometer under sub-urban wintry conditions. *J Aerosol Sci* 42, 859-866.
507
508 Aurell, J., and Gullett, B. K. 2013. Emission Factors from Aerial and Ground
509 Measurements of Field and Laboratory Forest Burns in the Southeastern US: PM_{2.5},
510 Black and Brown Carbon, VOC, and PCDD/PCDF. *Environmental Science &*
511 *Technology* 47, 8443-8452.

512 Benscoter, B. W., and Wieder, R. K. 2003. Variability in organic matter lost by
513 combustion in a boreal bog during the 2001 Chisholm fire. *Canadian Journal of Forest*
514 *Research* 33, 2509-2513.

515

516 Bertschi, I., Yokelson, R. J., Ward, D. E., Babbitt, R. E., Susott, R. A., Goode, J. G.,
517 and Hao, W. M. 2003. Trace gas and particle emissions from fires in large diameter
518 and belowground biomass fuels. *Journal of Geophysical Research: Atmospheres* 108,
519 8472.

520

521 Black, R. R., Meyer, C. P., Touati, A., Gullett, B. K., Fiedler, H., and Mueller, J. F.
522 2011. Emissions of PCDD and PCDF from combustion of forest fuels and sugarcane:
523 A comparison between field measurements and simulations in a laboratory burn
524 facility. *Chemosphere* 83, 1331-1338.

525

526 Black, R. R., Meyer, C. P., Touati, A., Gullett, B. K., Fiedler, H., and Mueller, J. F.
527 2012a. Emission factors for PCDD/PCDF and dl-PCB from open burning of biomass.
528 *Environ Int* 38, 62-66.

529

530 Black, R. R., Meyer, C. P., Yates, A., Van Zwieten, L., Chittim, B. G., Gaus, C., and
531 Mueller, J. F. 2012b. Release of native and mass labelled PCDD/PCDF from soil
532 heated to simulate bushfires. *Environ Pollut* 166, 10-16.

533

534 Black, R. R., Meyer, C. P., Yates, A., Van Zwieten, L., Chittim, B. G., and Mueller, J.
535 F. 2012c. Release of PCDD/PCDF to air and land during open burning of sugarcane
536 and forest litter over soil fortified with mass labelled PCDD/PCDF. *Atmos Environ*
537 59, 125-130.

538

539 Chen, L. W. A., Verburg, P., Shackelford, A., Zhu, D., Susfalk, R., Chow, J. C., and
540 Watson, J. G. 2010. Moisture effects on carbon and nitrogen emission from burning of
541 wildland biomass. *Atmos Chem Phys* 10, 6617-6625.

542

543 Christian, T. J., Kleiss, B., Yokelson, R. J., Holzinger, R., Crutzen, P. J., Hao, W. M.,
544 Saharjo, B. H., and Ward, D. E. 2003. Comprehensive laboratory measurements of
545 biomass-burning emissions: 1. Emissions from Indonesian, African, and other fuels.
546 *Journal of Geophysical Research: Atmospheres* 108, 4719.

547

548 Davies, G. M., Gray, A., Rein, G., and Legg, C. J. 2013. Peat consumption and carbon
549 loss due to smouldering wildfire in a temperate peatland. *Forest Ecol Manag* 308,
550 169-177.

551

552 Dhammapala, R., Claiborn, C., Jimenez, J., Corkill, J., Gullett, B., Simpson, C., and
553 Paulsen, M. 2007. Emission factors of PAHs, methoxyphenols, levoglucosan,
554 elemental carbon and organic carbon from simulated wheat and Kentucky bluegrass
555 stubble burns. *Atmos Environ* 41, 2660-2669.

556

557 EPA, U. S. 1996. EPA Test Method 0023a. . SW-846 (NTIS PB88-239223), Office of
558 Solid Waste and Emergency Response, US EPA: Washington, DC, 1996.

559 EPA, U. S. 1999. Compendium Method TO-9A, Determination Of Polychlorinated,
560 Polybrominated And Brominated/Chlorinated Dibenzo-p-Dioxins And Dibenzofurans

561 In Ambient Air. Compendium of Methods for the Determination of Toxic Organic
 562 Compounds in Ambient Air, 2nd Ed EPA/625/R-96/010b,
 563 <http://www.epa.gov/ttnamti1/files/ambient/airtox/to-9arr.pdf>, accessed April 24, 2015.
 564
 565 EPA, U. S. 2007. SW-846 Method 8270D, Semivolatile Organic Compounds by Gas
 566 Chromatography/Mass Spectrometry (GC/MS).
 567 <http://www.epa.gov/osw/hazard/testmethods/sw846/pdfs/8270d.pdf>, accessed April
 568 24, 2015.
 569
 570 George, I. B., R.; Geron, C.; Aurell, J.; Hays, M.; Preston, W.; Gullett, B. 2015.
 571 Speciated gas and particulate organic emissions from laboratory peat fires. Submitted
 572 to Atmospheric Environment.
 573
 574 Geron, C., and Hays, M. 2013. Air emissions from organic soil burning on the coastal
 575 plain of North Carolina. *Atmos Environ* 64, 192-199.
 576
 577 Grandesso, E., Gullett, B., Touati, A., and Tabor, D. 2011. Effect of Moisture, Charge
 578 Size, and Chlorine Concentration on PCDD/F Emissions from Simulated Open
 579 Burning of Forest Biomass. *Environmental Science & Technology* 45, 3887-3894.
 580
 581 Gullett, B., Touati, A., and Oudejans, L. 2008. PCDD/F and aromatic emissions from
 582 simulated forest and grassland fires. *Atmos Environ* 42, 7997-8006.
 583
 584 Hays, M. D., Geron, C. D., Linna, K. J., Smith, N. D., and Schauer, J. J. 2002.
 585 Speciation of gas-phase and fine particle emissions from burning of foliar fuels.
 586 *Environmental Science & Technology* 36, 2281-2295.
 587
 588 Hosseini, S., Urbanski, S. P., Dixit, P., Qi, L., Burling, I., Yokelson, R., Shrivastava,
 589 M., Jung, H., Weise, D. R., Miller, W., and Cocker, D. 2013. Laboratory
 590 characterization of PM emissions from combustion of wildland biomass fuels. *Journal*
 591 *of Geophysical Research: Atmospheres* 118, 9914-9929.
 592
 593 Inuma, Y., Brueggemann, E., Gnauk, T., Mueller, K., Andreae, M. O., Helas, G.,
 594 Parmar, R., and Herrmann, H. 2007. Source characterization of biomass burning
 595 particles: The combustion of selected European conifers, African hardwood, savanna
 596 grass, and German and Indonesian peat. *J Geophys Res-Atmos* 112.
 597
 598 Jenkins, B. M., Jones, A. D., Turn, S. Q., and Williams, R. B. 1996. Emission Factors
 599 for Polycyclic Aromatic Hydrocarbons from Biomass Burning. *Environmental*
 600 *Science & Technology* 30, 2462-2469.
 601
 602 Khan, B., Hays, M. D., Geron, C., and Jetter, J. 2012. Differences in the OC/EC
 603 Ratios that Characterize Ambient and Source Aerosols due to Thermal-Optical
 604 Analysis. *Aerosol Science and Technology* 46, 127-137.
 605
 606 Kim, Y. H., Tong, H., Daniels, M., Boykin, E., Krantz, Q. T., McGee, J., Hays, M.,
 607 Kovalcik, K., Dye, J. A., and Gilmour, M. I. 2014. Cardiopulmonary toxicity of peat
 608 wildfire particulate matter and the predictive utility of precision cut lung slices.
 609 *Particle and Fibre Toxicology* 11 (29).

610 Larsen, J. C. L., P.B. 1998. Chemical carcinogens. In: Air pollution and health.
 611 Cambridge, UK: The Royal Society of Chemistry, 33-56.
 612
 613 Liu, S., Aiken, A. C., Arata, C., Dubey, M. K., Stockwell, C. E., Yokelson, R. J.,
 614 Stone, E. A., Jayarathne, T., Robinson, A. L., DeMott, P. J., and Kreidenweis, S. M.
 615 2014. Aerosol single scattering albedo dependence on biomass combustion efficiency:
 616 Laboratory and field studies. *Geophys Res Lett* 41, 742-748.
 617
 618 May, A. A., McMeeking, G. R., Lee, T., Taylor, J. W., Craven, J. S., Burling, I.,
 619 Sullivan, A. P., Akagi, S., Collett, J. L., Jr., Flynn, M., *et al.* 2014. Aerosol emissions
 620 from prescribed fires in the United States: A synthesis of laboratory and aircraft
 621 measurements. *J Geophys Res-Atmos* 119, 11826-11849.
 622
 623 McMeeking, G. R., Kreidenweis, S. M., Baker, S., Carrico, C. M., Chow, J. C.,
 624 Collett, J. L., Hao, W. M., Holden, A. S., Kirchstetter, T. W., Malm, W. C., *et al.*
 625 2009. Emissions of trace gases and aerosols during the open combustion of biomass in
 626 the laboratory. *J Geophys Res-Atmos* 114.
 627
 628 Mickler, R., Rorig, M., and Geron, C. 2011. Project Title: Predicting Prescribed and
 629 Wildland Fire Smoke, Emissions, and Fire Characteristics in Deep Organic Soils.
 630 Final Report: Joint Fire Science Program Project Number: 08-1-3-03
 631 http://www.firescience.gov/projects/08-1-3-03/project/08-1-3-03_final_report.pdf,
 632 accessed April 24, 2015.
 633
 634 NTP 2014. National Toxicology Program, Polycyclic Aromatic Hydrocarbons: 15
 635 Listings. In: Report on Carcinogens, Thirteenth Edition, Research Triangle Park, NC:
 636 US Department of Health and Human Services, Public Health Service
 637 <http://ntpniehsnihgov/pubhealth/roc/roc13/> Accessed April 24, 2015.
 638
 639 Poulter, B., Christensen, N. L., and Halpin, P. N. 2006. Carbon emissions from a
 640 temperate peat fire and its relevance to interannual variability of trace atmospheric
 641 greenhouse gases. *J Geophys Res* 111.
 642
 643 Rappold, A. G., Stone, S. L., Cascio, W. E., Neas, L. M., Kilaru, V. J., Carraway, M.
 644 S., Szykman, J. J., Ising, A., Cleve, W. E., Meredith, J. T., *et al.* 2011. Peat Bog
 645 Wildfire Smoke Exposure in Rural North Carolina Is Associated with
 646 Cardiopulmonary Emergency Department Visits Assessed through Syndromic
 647 Surveillance. *Environ Health Persp* 119, 1415-1420.
 648
 649 Rein, G., Cohen, S., and Simeoni, A. 2009. Carbon emissions from smouldering peat
 650 in shallow and strong fronts. *P Combust Inst* 32, 2489-2496.
 651
 652 See, S. W., Balasubramanian, R., Rianawati, E., Karthikeyan, S., and Streets, D. G.
 653 2007. Characterization and source apportionment of particulate matter $\leq 2.5 \mu m$ in
 654 Sumatra, Indonesia, during a recent peat fire episode. *Environmental Science &*
 655 *Technology* 41, 3488-3494.
 656
 657 Stockwell, C. E., Yokelson, R. J., Kreidenweis, S. M., Robinson, A. L., DeMott, P. J.,
 658 Sullivan, R. C., Reardon, J., Ryan, K. C., Griffith, D. W. T., and Stevens, L. 2014.
 659 Trace gas emissions from combustion of peat, crop residue, domestic biofuels,

660 grasses, and other fuels: configuration and Fourier transform infrared (FTIR)
 661 component of the fourth Fire Lab at Missoula Experiment (FLAME-4). *Atmos Chem*
 662 *Phys* 14, 9727-9754.

663

664 Turetsky, M. R., Benscoter, B., Page, S., Rein, G., van der Werf, G. R., and Watts, A.
 665 2015. Global vulnerability of peatlands to fire and carbon loss. *Nat Geosci* 8, 11-14.

666 Turetsky, M. R., and Wieder, R. K. 2001. A direct approach to quantifying organic
 667 matter lost as a result of peatland wildfire. *Canadian Journal of Forest Research* 31,
 668 363-366.

669

670 Ward, D. E., Nelson, R. M., and Adams, D. F. 1979. Forest fire smoke plume
 671 documentation. Proceedings of the 77th Annual Meeting of the Air Pollution Control
 672 Association, Pap. 079-6.3, A&WMA, Pittsburgh, PA.

673

674 Ward, D. E., and Radke, L. F. 1993. Emissions measurements from vegetation fires:
 675 A comparative evaluation of methods and results, Eds. Crutzen, P. & Goldammer, J.
 676 *Environmental Sciences Research Report*, Vol 13, 53-76.

677

678 Woodall, B. D., Yamamoto, D. P., Touati, A., and Gullett, B. K. 2012.
 679 Characterization of Emissions from Simulated Deployed US Military Waste.
 680 *Environmental Science & Technology* 46, 10997-11003.

681

682 Yelverton, T., Hays, M., Gullett, B. K., and Linak, W. P. 2014. Black carbon
 683 measurements of flame-generated soot as determined by optical, thermal-optical,
 684 direct absorption, and laser incandescence methods. *Environ Eng Sci* 31, 209-215.

685

686 Yokelson, R. J., Susott, R., Ward, D. E., Reardon, J., and Griffith, D. W. T. 1997.
 687 Emissions from smoldering combustion of biomass measured by open-path Fourier
 688 transform infrared spectroscopy. *Journal of Geophysical Research: Atmospheres* 102,
 689 18865-18877.

690