



## New estimate of particulate emissions from Indonesian peat fires in 2015

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**Abstract.** Indonesia contains large areas of peatland that have been drained and cleared of natural vegetation, making them susceptible to burning. Peat fires emit considerable amounts of carbon dioxide, particulate matter (PM) and other trace gases, contributing to climate change and causing regional air pollution. However, emissions from peat fires are uncertain, due to uncertainties in emission factors and fuel consumption. We used the Weather Research and Forecasting model with chemistry and measurements of PM concentrations to constrain PM emissions from Indonesian fires during 2015, one of the largest fire seasons in recent decades. We estimate primary PM<sub>2.5</sub> (particles with diameters less than 2.5 μm) emissions from fires across Sumatra and Borneo during September–October 2015 were 7.33 Tg, a factor 3.5 greater than those in the Fire Inventory from NCAR (FINNv1.5), which does not include peat burning. We estimate similar dry fuel consumption and CO<sub>2</sub> emissions to those in the Global Fire Emissions Database (GFED4s, including small fires) but PM<sub>2.5</sub> emissions that are a factor of 1.8 greater, due to updated PM<sub>2.5</sub> emission factors for

Indonesian peat. Fires were responsible for an additional 3.12 Tg of secondary organic aerosol formation. Through comparing simulated and measured PM concentrations, our work provides independent support of these updated emission factors. We estimate peat burning contributed 71 % of total primary PM<sub>2.5</sub> emissions from fires in Indonesia during September–October 2015. We show that using satellite-retrieved soil moisture to modify the assumed depth of peat burn improves the simulation of PM, increasing the correlation between simulated and observed PM from 0.48 to 0.56. Overall, our work suggests that peat fires in Indonesia produce substantially greater PM emissions than estimated in current emission inventories, with implications for the predicted air quality impacts of peat burning.

## 1 Introduction

Vegetation and peatland fires across Indonesia and Malaysia result in habitat and biodiversity loss, large emissions of carbon and regional haze episodes. Fire events cause regional reductions in visibility and severe air pollution (Reddington et al., 2014; Gaveau et al., 2014b; Kim et al., 2015; Lee et al., 2017) with associated morbidity and mortality (Crippa et al., 2016; Marlier et al., 2012; Reddington et al., 2015).

Indonesia contains 36 % of the world's tropical peatland, the largest of any country in the tropics (Dargie et al., 2017; Page et al., 2011). Undisturbed peatlands typically have high moisture content, making them naturally resilient to fire (Wösten et al., 2008). Indonesian peatlands are experiencing deforestation and conversion to agriculture, oil palm and timber plantations (Hansen et al., 2013; Gaveau et al., 2014a; Miettinen et al., 2017). During this conversion, drainage canals are installed, lowering the water table and making the peatland more susceptible to burning (Konecny et al., 2016). Fire is also used as an agricultural tool to clear vegetation (Carlson et al., 2012; Page et al., 2002). These human disturbances can make peatlands particularly prone to fire. In 2015, 53 % of fires in Indonesia occurred on peatland, which made up only 12 % of the land area (Miettinen et al., 2017).

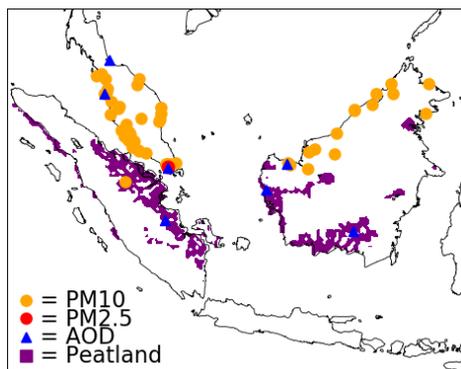
Peatlands have thick organic soil layers up to 10 m deep (Hu et al., 2018). Fires on peatland can burn into these underground organic layers and smoulder for weeks after the surface fire has gone out (Roulston et al., 2018), resulting in substantially greater emissions compared to surface vegetation fires (Heil et al., 2007). Peat fires are estimated to contribute 3.7 % of global fire carbon emissions (van der Werf et al., 2017). In Indonesia, peatland fires are the largest contributor to fire emissions in the region (Reddington et al., 2014; van der Werf et al., 2010). For the fires in 2015, Wooster et al. (2018) found that 95 % of the particulate matter (PM<sub>2.5</sub>) emissions came from peatland fires, and Wiggins et al. (2018) estimated that 85 % of smoke plumes detected in Singapore originated from peat fires.

Whilst it is known that emissions from peatland fires are substantial, current emissions estimates have large uncertainties. Emission estimates are typically based on remotely sensed information from satellites on the area burned by the fires. Burned area may be underestimated in Southeast Asia due to extensive cloud cover (Ge et al., 2014). Furthermore, estimates of burned area are limited to surface fires and may miss fires that burn underground (Kaiser et al., 2012). For peat fires, the amount of biomass consumed by the fire depends on how deep into the peat the fire burns (Hu et al., 2018). Burn depth is variable, with some fires recorded as burning to a depth of 0.85 m, resulting in carbon emissions of 31.5 kg C m<sup>-2</sup> (Page et al., 2002). Burn depth depends on the level of the water table and the water content of the peat, with increased burn depth when the water table is lowered and the peat dries out (Ballhorn et al., 2009; Huang and Rein, 2015; Rein et al., 2008). Konecny et al. (2016) also

suggest that burn depth changes based on the frequency of fire, with reduced burn depth for repeat fires at the same location. Information on the spatial and temporal variability of burn depth is limited and current emission inventories make broad assumptions regarding these parameters. Emission factors (EFs), estimated from field or laboratory measurements, are used to convert mass of fuel consumed by the fire to the emitted mass of gas-phase and particulate pollutants (e.g. Andreae and Merlet, 2001; Akagi et al., 2011). Compared to flaming combustion, smouldering peat fires have colder combustion temperatures and typically higher EFs for products of incomplete combustion including CO, CH<sub>4</sub>, CO<sub>2</sub>, HCN, NH<sub>3</sub> and particulate matter (PM; Stockwell et al., 2016). Until recently, there have been few specific measurements of EFs for tropical peat fires. Roulston et al. (2018) and Wooster et al. (2018) found that EFs for tropical peat fires could be underestimated by a factor of three (PM<sub>2.5</sub> EF from peat fires is assumed to be 9.1 g kg<sup>-1</sup> in the Global Fire Emissions Database (GFED4), compared to 24 g kg<sup>-1</sup> suggested by Roulston et al., 2018, and 28 g kg<sup>-1</sup> suggested by Wooster et al., 2018). There are large variations in EFs for peat in Indonesia. In one study measuring emissions from peat fires in central Kalimantan during 7 d in 2015, PM<sub>2.5</sub> EFs were found to vary between 6 and 30 g kg<sup>-1</sup> (Jayarathne et al., 2018). Kuwata et al. (2018) used measurements from Indonesian peatland fires to estimate EFs of PM<sub>10</sub> of 13 ± 2 g kg<sup>-1</sup> in 2013 and 19 ± 2 g kg<sup>-1</sup> in 2014.

These uncertainties cause corresponding uncertainty in estimates of emissions from peat fires and impacts on the regional air pollution. Previous studies underestimate measured and aerosol optical depth (AOD) or PM and scale particulate fire emissions from global fire emissions inventories or simulated fire-derived aerosol by a factor of 1.36–3.00 in order to match observations (Johnston et al., 2012; Koplitz et al., 2016; Marlier et al., 2012; Reddington et al., 2016; Tosca et al., 2013; Ward et al., 2012). This suggests that particulate emissions from tropical peatland regions are underestimated in current fire emission inventories.

Severe fire events in Indonesia occur during periods of drought (van der Werf et al., 2008; Tosca et al., 2011; Gaveau et al., 2014b; Field et al., 2016), resulting in strong seasonal and interannual variability. Severe droughts lower the water table, exposing more peat and increasing susceptibility to burning. Extensive fires and regional haze episodes across Indonesia occurred in 1982–1983, 1997–1998, 2006, 2009, 2013 and 2015. During September–October 2015, dry conditions caused by a strong El Niño resulted in large fires across Sumatra and Kalimantan. This fire episode was the largest in Indonesia since 1997 (Huijnen et al., 2016), releasing an estimated 188 ± 67 Tg C (Huijnen et al., 2016) as CO<sub>2</sub> and 149 ± 71 Tg C from peat fires (Jayarathne et al., 2018). The fires also emitted substantial amounts of PM<sub>2.5</sub>, estimated at 9.1 ± 3.2 Tg (Wooster et al., 2018), with 6.5 ± 5.5 Tg from peat fires (Jayarathne et al., 2018). Particulate air pollution from these fires may have caused between 6513 and 17 270



**Figure 1.** The study area showing the locations of PM<sub>10</sub> measurements in yellow circles, PM<sub>2.5</sub> in red circles and AOD in blue triangles. Peatland is shown in purple.

excess premature deaths through short-term exposure to fire-sourced PM<sub>2.5</sub> (Crippa et al., 2016) and as many as 100 300 excess premature deaths over the longer term due to exposure to this pollution (Kopplitz et al., 2016).

Given the importance of peatland fires as the main contributor to fire emissions in Indonesia, there is a high priority in reducing the large uncertainties in these emissions. In this study we aim to improve our understanding of the emissions from peat fires in Indonesia by combining fire emission inventories, a regional air quality model and extensive measurements of PM. We focus on the large fires of September–October 2015. We updated an existing fire emissions inventory to include emissions from peat fires, applying updated information on emission factors from tropical peat combustion and using satellite-retrieved information on soil moisture to control assumed depth of peat burn. We used the existing and new emissions inventories with an air quality model and evaluated simulated PM concentrations against observations. The new emissions inventory demonstrates a substantial improvement in simulating regional PM<sub>2.5</sub> concentrations.

## 2 Methodology

We used a regional atmospheric model to simulate PM concentrations during August–October 2015, with different combinations of peat and vegetation fire emissions, described below. Our study region included Borneo, Sumatra and mainland Malaysia (Fig. 1, 10° S–10° N and 95–120° E), which is at the centre of the model domain. We used surface observations of PM and AOD to assess the performance of the model with the different fire emissions. Atmospheric PM concentrations are impacted by a range of atmospheric processes, including atmospheric transport and deposition and secondary production of aerosol. Evaluating the fire emissions is complicated by the treatment of these processes in the model.

### 2.1 WRF-chem model

We used the Weather Research and Forecasting model with Chemistry (WRF-chem), version 3.7.1. WRF-chem simulates gas-phase chemistry and aerosol processes fully coupled to the meteorology (Grell et al., 2005). The model was run at 30 km horizontal resolution with 33 vertical levels over 140 × 140 grid points centred at 0° N, 110° E (17° S–18° N, 90–130° E), using the Mercator projection. Simulations were run over the period from 18 July until 1 November. The MOZART (Model for Ozone and Related Chemical Tracers, version 4; Emmons et al., 2010) chemistry scheme was used to calculate gas-phase chemical reactions, with aerosol dynamics and processes represented by MOSAIC (Model for Simulating Aerosol Interactions and Chemistry; Zaveri et al., 2008; Hodzic and Knote, 2014). This includes a secondary organic aerosol (SOA) scheme based on Hodzic and Jimenez (2011). Primary organic aerosols (POAs) are considered non-volatile in the model. Within MOSAIC, four aerosol bin sizes were used: 0.039–0.156, 0.156–0.625, 0.625–2.5 and 2.5–10 μm. Anthropogenic emissions were from EDGAR-HTAP2 (Janssens-Maenhout et al., 2015) for 2010, and biogenic emissions were from MEGAN (Model of Emissions of Gases and Aerosols from Nature; Guenther et al., 2006). A similar model setup has been used for studies in India (Conibear et al., 2018), the United States (Knote et al., 2014) and Indonesia (Crippa et al., 2016). The meteorology was re-initialized each month with NCEP GFS (NCEP, 2007), with a 24 h spin-up, and was then free running for the rest of the month. More information on the chemistry and physics options used can be found in Table S1 in the Supplement.

### 2.2 Fire emissions

We applied four different emission inventories in the WRF-chem model, all based on the Fire Inventory from NCAR (FINNv1.5). All emission inventories included emissions from vegetation fires as treated in FINNv1.5 but with different treatment of peat combustion, as described below.

#### 1. FINN (FINN).

The Fire Inventory from NCAR (FINNv1.5) combines data on active fires, biomass burned and EFs to give daily fire emissions at 1 km resolution (Wiedinmyer et al., 2011). Vegetation burned is assigned based on the MODIS Land Cover Type and Vegetation Continuous Field (VCF) products. Fire area burned is assumed to be 1 km<sup>2</sup> (100 ha) per hotspot detected (scaled back by any non-vegetated area assigned by the VCF product). Fuel loading is from Hoelzemann et al. (2004) and EFs are from Akagi et al. (2011), Andreae and Merlet (2001), and McMeeking et al. (2009). FINNv1.5 includes emissions from combustion of above-ground vegetation but does not include emissions from combustion of peat.

### 2. FINN with GFED4s peat (FINN+GFEDpeat).

In this inventory we combined vegetation emissions from FINNv1.5 with emissions from peat fires from the Global Fire Emissions Database (version 4, including small fires; GFED4s). GFED combines burned area from Giglio et al. (2013) with assumed combustion completeness and EFs. For peat fires the depth burned is dependent on the soil moisture, with a maximum depth of 0.5 m. GFED4s peat EFs come from studies on Indonesian peat fires for CO<sub>2</sub>, CO and CH<sub>4</sub> and from deforestation fires for all other species.

GFED4s data are available daily at 0.25° resolution. GFED emissions are available split by fuel type, allowing us to combine GFED4s emissions from peat fires with FINN emissions from other fuel types.

### 3. FINN with peat emissions (FINNpeat).

We created a new emissions inventory (FINNpeat), based on FINNv1.5 emissions with the addition of emissions from combustion of peat. Emissions from vegetation fires in FINNpeat are identical to those in FINN. For those fire detections occurring on peat, as identified using a peatland distribution map (WRI), additional emissions from the peat burning were calculated using Eq. (1):

$$E_s = BA \times BD \times \rho \times EF_s, \quad (1)$$

where  $E_s$  is the emissions of a species,  $s$ , from a fire, BA is the burned area and BD is the burn depth for the fire,  $\rho$  is the peat density, and  $EF_s$  is the emissions factor for species,  $s$ . The peatland map only includes peatlands in Indonesia, so emissions from Malaysian peat fires are not included. For each fire, the corresponding emissions are released on the day that the fire was detected, with no long-term smouldering effects, which may be important for peat fires.

Tansey et al. (2008) used an analysis of MODIS hotspots and MODIS burned area in a peat swamp in Indonesia to estimate 15–16 ha of burned area per hotspot. However, 60% of burned areas did not have an identified hotspot, implying an area burned per MODIS hotspot of approximately 40 ha. Over areas defined as peatland, we therefore assumed a burned area of 40 ha of peat burnt per hot spot, smaller than the 100 ha assumed for vegetation fires.

The mass of peat burned during peat fires was calculated from an assigned burned area, peat density and burn depth (Table 1). We assumed a peat density of 0.11 g cm<sup>-3</sup> (Driessen and Rochimah, 1976; Neuzil, 1997; Shimada et al., 2001; Warren et al., 2012) and a burn depth of 37 cm for all fires detected (Page et al., 2002; Usup et al., 2004; Ballhorn et al., 2009). We assumed that all peat within the burned area and depth is

combusted, as is assumed in GFED3 (van der Werf et al., 2006). This gives a fuel consumption of 40.7 kg dry matter m<sup>-2</sup>, consistent with van Leeuwen et al. (2014), who found the average fuel consumption for Indonesian peatland fire to be 31.4 kg dry matter m<sup>-2</sup> (from studies by Page et al., 2002; Usup et al., 2004; Ballhorn et al., 2009).

We assigned the average EFs from previous studies (Table 2) (Christian et al., 2003; Hatch et al., 2015; Stockwell et al., 2016; Wooster et al., 2018; Jayarathne et al., 2018; Nara et al., 2017; Smith et al., 2018): CO<sub>2</sub> (1670 g kg<sup>-1</sup>), PM<sub>2.5</sub> (22.3 g kg<sup>-1</sup>), organic carbon (OC) (11.5 g kg<sup>-1</sup>) and black carbon (BC) (0.07 g kg<sup>-1</sup>). By comparison, GFED4s assumes similar EFs for CO<sub>2</sub> (1703 g kg<sup>-1</sup>) and BC (0.04 g kg<sup>-1</sup>) but substantially lower EFs for PM<sub>2.5</sub> (9.1 g kg<sup>-1</sup>) and OC (6.02 g kg<sup>-1</sup>). Some of the EFs used have been calculated from fires on peatland that also contain vegetation burning.

The variation in measured EFs vary widely depending on the emitted pollutant, 20% for CO<sub>2</sub> (1507–1775 g kg<sup>-1</sup>), a factor 2–3 for PM<sub>2.5</sub> (17.3–28.0 g kg<sup>-1</sup>) and OC (6.02–16.0 g kg<sup>-1</sup>) and an order of magnitude for BC (0.006–0.134 g kg<sup>-1</sup>). The EFs used by Wooster et al. (2018) for PM<sub>2.5</sub> and CO<sub>2</sub> are at the upper end of the ranges of EFs considered for this study. Substantial uncertainty in BC emissions has implications for the climate impacts of the aerosol, but since BC only makes a minor contribution to overall mass it has less importance for simulation of PM<sub>2.5</sub>.

### 4. FINN new peat with soil moisture (FINNpeatSM).

As peat dries out, the burn depth increases (Rein et al., 2008; Usup et al., 2004; Wösten et al., 2008). However, FINNpeat assumes a constant peat burn of 37 cm depth regardless of soil moisture. FINNpeatSM emissions were calculated in the same way as FINNpeat emissions but with peat burn depth varying depending on surface soil moisture.

Daily soil moisture from the European Space Agency (ESA CCI SMv04.4) was used to estimate the burn depth of peat (Dorigo et al., 2017; Gruber et al., 2017; Liu et al., 2012). Frequent cloud cover leads to numerous missing values in the daily soil moisture data at 0.25° resolution. To help account for this, soil moisture was averaged to 2° resolution. In 2015, average daily soil moisture across peatlands in the study area declined from around 0.24 m<sup>3</sup> m<sup>-3</sup> in August, to 0.23 m<sup>3</sup> m<sup>-3</sup> in September to a minimum of around 0.22 m<sup>3</sup> m<sup>-3</sup> in October 2015, then increasing to 0.25 m<sup>3</sup> m<sup>-3</sup> in November (Fig. S1). By comparing the temporal change in soil moisture over high fire regions in Sumatra and Kalimantan, we chose upper and lower limits of 0.25 and

**Table 1.** Values for peat burn depth and peat density found in previous studies and the average value across studies. All studies were based in Kalimantan, Indonesia.

	Burn depth (m)	Peat density (g cm <sup>-3</sup> )
Page et al. (2002)	0.51	
Ballhorn et al. (2009)	0.33	
Centre for international co-operation in measurement of tropical peatlands (From Ballhorn et al., 2009)	0.3	
Usup et al. (2004)	0.35	
Stockwell et al. (2016)	0.34	
Neuzil (1997)		0.093
Driessen and Rochimah (1976)		0.11
Warren et al. (2012)		0.127
Shimada et al. (2001)		0.112
Konecny et al. (2016)		0.121
Average	0.37	0.11

0.15 m<sup>3</sup> m<sup>-3</sup>, which reflected the soil moisture in these regions before and during the dry season (Fig. S2).

We scaled burn depth linearly from a minimum of 5 cm for a soil moisture of 0.25 m<sup>3</sup> m<sup>-3</sup> to a maximum of 37 cm for soil moisture of 0.15 m<sup>3</sup> m<sup>-3</sup>. Under these assumptions, mean peat burn depth across peatland areas in Indonesia increased from 15.0 cm in August to 23.6 cm in September and 24.8 cm in October.

The PM<sub>2.5</sub> EF used for FINNpeat and FINNpeatSM is at the higher end of the range of values used in other studies. The same emissions combined with a lower EF would require a greater burn depth or area burned per fire hotspot.

### Vertical profile of fire emissions

Fires can inject emissions into air above the surface layer of the model, which in this model setup is about 70 m. By default in WRF-chem, the vertical distribution of fire emissions uses a plume rise parameterization based on a 1d cloud model (Freitas et al., 2007). However, smoke from smouldering peat fires can be emitted close to the ground. Recent work suggests that tropical fires mostly inject emissions into the boundary layer (BL) and the WRF-chem scheme may overestimate fire injection heights. Tosca et al. (2011) found that the average plume height for fires in Sumatra and Borneo was 729 m, with 96 % of plumes confined to within 500 m of the boundary layer. Martin et al. (2018) found that 90 % of fire emissions in South Asia in September–November were injected below 1500 m. Archer-Nicholls et al. (2015) found that the WRF-chem plume rise parameterization overestimated the injection height for fires in South America. For this reason we chose not to use the plume injection option and in-

stead tested two alternate approaches to control the vertical profile of fire emissions:

- All of the emissions were added to the surface model layer (surface injection).
- Half of the emissions were added to the surface model layer and the other half of the emissions were spread evenly into model layers throughout the boundary layer (boundary layer injection).

### 2.3 Particulate measurements

Measurements of particulate matter with a diameter of less than 2.5 μm (PM<sub>2.5</sub>), less than 10 μm (PM<sub>10</sub>) and less than 1 μm (PM<sub>1</sub>) and measurements of AOD (Table 3) were used to evaluate the model. Figure 1 shows the locations of measurements. Hourly measurements of PM<sub>2.5</sub> concentrations are available from the National Environment Agency of Singapore for five sites in Singapore during October 2015. We averaged concentrations across the five sites to produce mean PM<sub>2.5</sub> concentrations for Singapore. From Singapore, there are also measurements of non-refractory, composition-resolved sub-micron PM from an aerosol chemical speciation monitor (ACSM) (Budisulistiorini et al., 2018). We summed the chemically resolved masses to give PM<sub>1</sub>. Betha et al. (2014) found that for fire-induced haze in Singapore in 2013, 96 %–99 % of the PM<sub>2.5</sub> was PM<sub>1</sub>.

In October 2015, measured PM<sub>1</sub> agreed to within 20 % of the mean PM<sub>2.5</sub> concentration from the NEA. Ground-based AOD measurements were available from seven Aerosol Robotic Network (AERONET) sites for August–October. Measurements of hourly PM<sub>10</sub> were available from 52 locations across Malaysia (Mead et al., 2018) and 1 location in Indonesia during August–October 2015. We compared daily mean observations at each site with simulated PM and AOD in Sect. 3.2. The fractional bias (defined in the Supplement) and correlation coefficients were used to evaluate the simulations. We did not use AOD data from MODIS retrievals, which significantly underestimated AOD over the region during this period, due to excluding smoke plumes that were mistaken for clouds (Shi et al., 2019).

## 3 Results

### 3.1 Fire emissions

Table 4 shows total dry matter consumption, PM<sub>2.5</sub>, CO<sub>2</sub>, CO and SOA emissions from fires across Sumatra and Borneo in September–October 2015. The dry fuel consumption is lowest for FINN (230 Tg), which does not include peat fires. Dry matter consumption is similar for GFED, FINN+GFEDpeat and FINNpeatSM (455, 514, 465 Tg, respectively) and is highest for FINNpeat (612 Tg). This is likely due to the peat burn depth being greatest for FINNpeat. Wooster et al. (2018)

**Table 2.** Emission factors from previous studies ( $\text{g kg}^{-1}$ ) and an average value across all studies.

	Christian et al. (2003)	Wooster et al. (2018)	Stockwell et al. (2016) <sup>b</sup>	Stockwell et al. (2015)	Hatch et al. (2015)	Jayarathne et al. (2018)	Smith et al. (2018)	Nara et al. (2017)	Average
Method	Lab	In situ	In situ	Lab	Lab	In situ	In situ	In situ	
CO <sub>2</sub>	1703	1775	1564	1507.23			1579	1663	1669
CO	210.3	279	291	224.66			251	205	243.48
CH <sub>4</sub>	20.80	7.9	9.51	11.69			11.00	7.6	11.17
C <sub>2</sub> H <sub>2</sub>	0.06		0.12	0.1644			0.06		0.11
C <sub>2</sub> H <sub>4</sub>	2.57		0.96	1.09			2.30		1.60
C <sub>5</sub> H <sub>8</sub>			0.0528	1.1382					0.5823
CH <sub>3</sub> OH	8.23		2.14	3.78					4.48
HCHO	1.40		0.867	1.532			0.77		1.220
C <sub>2</sub> H <sub>4</sub> O <sub>2</sub>	1.59		0.108						0.849
CH <sub>3</sub> CHO	3.27		0.697	1.496					1.740
HCOOH	0.79		0.18	0.53			0.25		0.46
C <sub>3</sub> H <sub>6</sub> O	1.5		0.69	1.38					1.18
C <sub>2</sub> H <sub>6</sub>			1.52						1.52
C <sub>3</sub> H <sub>8</sub>			0.989						0.989
C <sub>10</sub> H <sub>16</sub>			0.00167	0.1925	0.0068				0.0984
NH <sub>3</sub>	19.92		2.86	1.33			7.82		7.09
PM <sub>2.5</sub>		28.0 <sup>a</sup>	21.5			17.3			22.3
Black carbon		0.134	0.00552						0.0695
Organic carbon	6.02		16.0			12.4			11.5
Higher alkanes			0.87						0.87

<sup>a</sup> Contains both peat and vegetation burning. <sup>b</sup> Stockwell et al. (2016) and Jayarathne et al. (2018) are based on the same measurements.

**Table 3.** Observational data for 2015.

Data	Location	Time period	Frequency of observations	Method	Reference/source
PM <sub>2.5</sub>	5 Sites in Singapore: N: 1.41° N, 103.80° E; E: 1.33° N, 103.69° E; C: 1.34° N, 103.82° E; W: 1.33° N, 103.93° E; S: 1.27° N, 103.83° E	28 September– 15 November 2015	1 h	Thermo Scientific™ 5030 SHARP monitor	National Environment Agency for Singapore
Composition-resolved non-refractory PM <sub>1</sub>	National Technological University in Singapore 1.35° N, 103.68° E	10–31 October 2015	2–3 min	Aerosol chemical speciation monitor (ACSM)	Budisulistiorini et al. (2018)
PM <sub>10</sub>	Pekanbaru, Indonesia 0.52° N, 101.43° E	1 January 2010 –31 December 2015	30 min	Measured using a Met One BAM 1020 real-time portable beta-attenuation mass monitor (BAM-1020)	
PM <sub>10</sub>	52 locations across Malaysia	August– November 2015	1 h	Measured using a Met One BAM 1020 real-time portable beta-attenuation mass monitor (BAM-1020)	Mead et al. (2018)
AOD	8 AERONET sites	August– October 2015	24 h average	Ground-based remote-sensing sun photometer instrument, measuring the intensity of solar radiation at 500 nm wavelength, from which AOD is derived	AERONET version 2

estimated  $358 \pm 107$  Tg of dry matter consumption for Kalimantan and Sumatra in September–October using satellite CO emissions (from the MOPITT instrument) and a CO EF. This dry matter estimate is in reasonable agreement with GFED and FINNpeatSM. FINN estimates a smaller dry matter consumption compared to Wooster et al. (2018), whereas FINNpeat estimates greater dry matter consumption. Whitburn et al. (2016), have estimated dry matter fuel consumption of 525 Tg, calculated using satellite CO emissions (from the IASI instrument) from peatlands and the CO EF for peat from GFAS. This estimate is larger than that found for GFED or FINNpeatSM and smaller than that found for FINNpeat.

Total September–October 2015 emissions of CO<sub>2</sub> follow a similar pattern to dry matter consumption, with similar values for GFED, FINN+GFEDpeat and FINNpeatSM (773, 822 and 781 Tg). The largest emissions are for FINNpeat (1014 Tg) and the smallest emissions for FINN (353 Tg). CO<sub>2</sub> EFs are similar for GFED and FINNpeat (1703 and 1669 g kg<sup>-1</sup>), explaining the similarity between dry matter consumption and CO<sub>2</sub> emissions for these inventories. The total CO<sub>2</sub> emissions for September–October estimated by Wooster et al. (2018) were  $692 \pm 213$  Tg, matching GFED, FINN+GFEDpeat and FINNpeatSM. Jayarathne et al. (2018) estimated  $547 \pm 259$  Tg of CO<sub>2</sub> were emitted from peat fires over southern Sumatra and Kalimantan, a range which includes the total CO<sub>2</sub> emissions from peat fires for FINN+GFEDpeat (469 Tg), FINNpeat (661 Tg) and FINNpeatSM (428 Tg).

Total emissions of PM<sub>2.5</sub> vary across simulations due to differences in assumed PM<sub>2.5</sub> EFs. FINN has the smallest total PM<sub>2.5</sub> emissions for September–October (2.09 Tg; Table 4). GFED and FINN+GFEDpeat have similar total PM<sub>2.5</sub> emissions (4.14 and 4.60 Tg), smaller than those for FINNpeatSM (7.33 Tg) despite these inventories having similar dry matter consumption and CO<sub>2</sub> emissions. This is due mainly to the difference in the assumed EFs for PM<sub>2.5</sub> from peat fires, with 9.1 g kg<sup>-1</sup> used in GFED and 22.26 g kg<sup>-1</sup> used in FINNpeatSM. Wooster et al. (2018), assumed a PM<sub>2.5</sub> EF of  $28 \pm 6$  g kg<sup>-1</sup> and estimated that  $9.1 \pm 3.2$  Tg of PM<sub>2.5</sub> was emitted over the whole of Sumatra and Kalimantan for September–October 2015, similar to that found in FINNpeat (10.60 Tg) and FINNpeatSM (7.33 Tg). In contrast, FINN and FINN+GFED, which use the lower EF, produce smaller PM<sub>2.5</sub> emissions by a factor of 2 and 4, respectively. Jayarathne et al. (2018) found that, for a smaller study area in Sumatra and Kalimantan, the total PM<sub>2.5</sub> emission from peat fires was  $6 \pm 5.5$  Tg, a range which covers the total PM<sub>2.5</sub> emissions from peat fires from FINN+GFEDpeat (2.51 Tg), FINNpeat (8.51 Tg) and FINNpeatSM (5.24 Tg).

GFED, FINN+GFED and FINNpeatSM all emit similar total amounts of CO over September–October (75, 77 and 78 Tg, respectively). This is likely due to the similar EFs used for peat fires (243 g kg<sup>-1</sup> in FINNpeatSM, 210 g kg<sup>-1</sup> in GFED). The total CO emissions from FINN are smaller (20 Tg) and are larger from FINNpeat (109 Tg). The contri-

bution from peat fires to total CO emissions (74 %–82 %) is larger than for CO<sub>2</sub> (55 %–65 %) and PM<sub>2.5</sub> (55 %–80 %). For every 1 g of CO emitted from fires, 0.04 g of SOA is assumed, and the total SOA from each fire emission inventory is shown in Table 4. The SOA increases the total PM emitted from fires to 2.89 Tg from FINN, 7.14 Tg from GFED, 7.68 Tg from FINN+GFED, 14.96 Tg from FINNpeat and 10.45 Tg from FINNpeatSM.

Table 4 also shows the fraction of emissions that are estimated to come from peat fires. Across the inventories that include peat burning, peat fires contribute 51 %–62 % of dry matter consumption, 55 %–65 % of CO<sub>2</sub> emissions and 55 %–80 % of PM<sub>2.5</sub> emissions. The emission inventories with updated PM<sub>2.5</sub> emission factors result in a greater contribution from peat burning (71 %–80 %) compared to emission inventories with the older EFs (55 %–62 %). Wooster et al. (2018) found that peatland fires contributed 85 % of the dry matter fuel consumption and 95 % of the PM<sub>2.5</sub> emissions in September–October 2015, greater than our estimates with updated EFs.

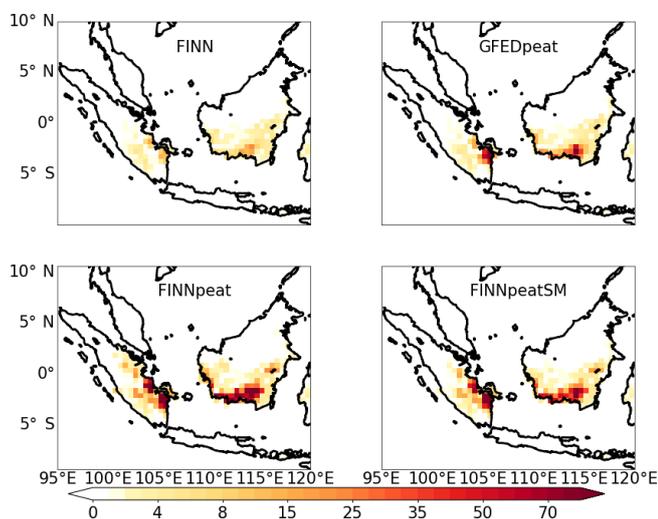
Figure 2 shows the spatial variations in the total PM<sub>2.5</sub> emissions during September–October 2015. In all inventories, the greatest emissions occur in southern Kalimantan and central and southern Sumatra, matching the locations of peatlands (Fig. 1). For the FINNpeatSM emissions, Sumatra contributes 42 % of the total PM<sub>2.5</sub> emissions; for FINNpeat, FINN+GFEDpeat and FINN, the contribution is 39 %, 40 % and 32 %, respectively. Wooster et al. (2018) found that 33 % of the total PM<sub>2.5</sub> emissions came from Sumatra, while Kopplitz et al. (2016) found that 47 % of OC and BC emitted in June to October 2015 came from Sumatra. Our estimates exclude fire emissions from eastern Indonesia. Nechita-banda et al. (2018) estimated that fires in eastern Indonesia and Papua New Guinea contributed around 15 %–20 % of total CO emissions from fires across the region, highlighting the need for future work to quantify PM emissions in this region.

FINN and GFED underestimate total emitted PM<sub>2.5</sub> and FINN underestimates total emitted CO<sub>2</sub> compared to the emissions found by Wooster et al. (2018) and Jayarathne et al. (2018), suggesting that peat fires are important contributors to these emissions. FINNpeatSM is the only emissions inventory that is consistent with these previous studies for both PM<sub>2.5</sub> and CO<sub>2</sub>.

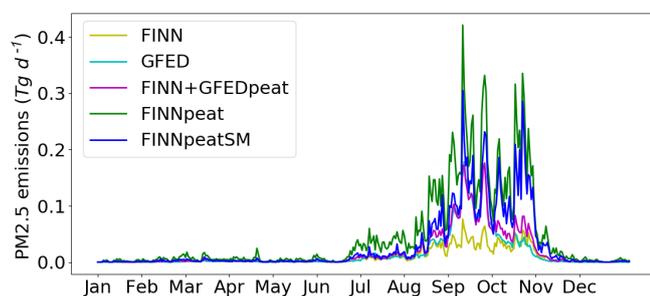
Figure 3 shows daily total PM<sub>2.5</sub> emissions from the different inventories over the study area. Temporally, the inventories follow a similar pattern, with 80 %–90 % of the total PM<sub>2.5</sub> emissions for 2015 occurring in August–October. For all the emissions inventories the majority of emissions are in September, followed by October and then August. GFED has the largest difference between September and October emissions (58 % in September and 17 % in October), followed by FINN+GFEDpeat (47 % and 24 %), FINNpeat (36 % and 30 %) and finally FINN (33 % and 29 %) and FINNpeatSM (36 % and 32 %), which have the smallest differences between the 2 months. The reduced ratio of the fraction of emis-

**Table 4.** Total dry matter fuel consumption, PM<sub>2.5</sub>, CO<sub>2</sub>, CO and SOA fire emissions for September–October 2015. Totals are for the area shown in Fig. 1. The percentage contribution from peat fires is indicated.

	FINN	GFED	FINN+GFEDpeat	FINNpeat	FINNpeatSM
Peat fires included	No	Yes	Yes	Yes	Yes
Dry matter fuel consumption (Tg)	230	455	514	612	465
CO <sub>2</sub> emissions (Tg)	353	773	822	1014	781
Contribution from peat fires	0 %	63 %	57 %	65 %	55 %
PM <sub>2.5</sub> emissions (Tg)	2.09	4.14	4.60	10.60	7.33
Contribution from peat fires	0 %	62 %	55 %	80 %	71 %
CO emissions (Tg)	20	75	77	109	78
Contribution from peat fires	0 %	80 %	74 %	82 %	74 %
SOA from biomass burning (Tg)	0.80	3.00	3.08	4.36	3.12



**Figure 2.** Total PM<sub>2.5</sub> fire emissions during September–October 2015 ( $\text{g m}^{-2}$ ).



**Figure 3.** Total daily PM<sub>2.5</sub> emissions from fires during 2015. Total shown for the area in Fig. 1: 10° S–10° N, 95–120° E.

sions in September compared to October for FINNpeatSM is due to greater soil moisture in September resulting in a reduced peat burn depth.

Another commonly used emissions inventory is the Global Fire Assimilation System (GFAS1). This uses satellite fire radiative power to detect fires, combined with EFs to calculate

daily emissions (Kaiser et al., 2012). For peat fires, some EFs are from studies of Indonesian peat, although the PM<sub>2.5</sub> EF ( $9.1 \text{ g kg}^{-1}$ ) is from tropical vegetation, as is used in GFED. Reddington et al. (2016) found that GFAS1 requires the same scaling as GFEDv3 to match observations in Indonesia. It is therefore likely that GFAS1 would show similar results to GFED in our assessment.

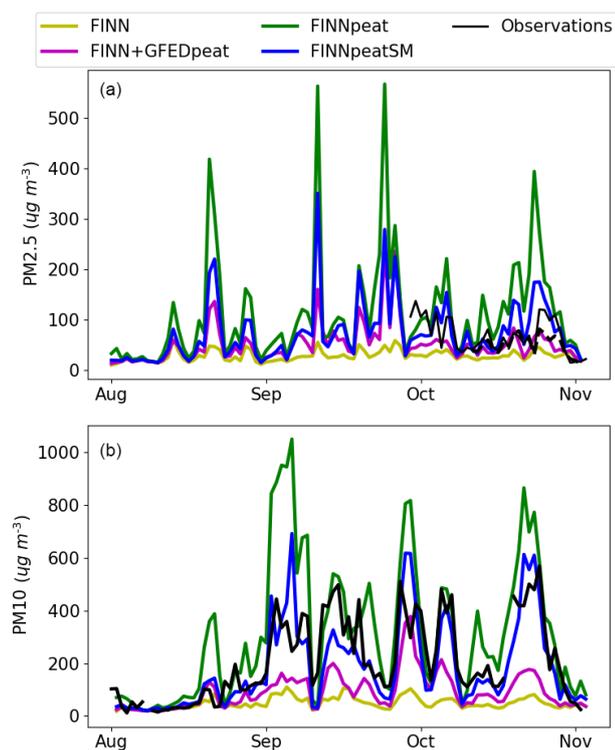
### 3.2 Comparison of model and observational data

We evaluated the WRF-chem simulations with the different emissions inventories and injection options against measured PM<sub>10</sub>, PM<sub>2.5</sub> and PM<sub>1</sub> concentrations. Figures 4 and 5 show the comparison of simulated and observed PM concentrations. Comparisons of PM<sub>2.5</sub> and PM<sub>1</sub> measurements only, which were restricted to Singapore, are shown in Fig. S3.

PM concentrations are underestimated by the model with FINN emissions, with a fractional bias (FB) of  $-0.67$  with surface injection and  $-0.77$  with boundary layer injection of emissions with an average across both simulations of  $-0.72$ . The model with FINN+GFEDpeat emissions also underestimates PM concentrations (average FB =  $-0.35$ ), whilst the model with FINNpeat emissions overestimates PM concentrations (average FB =  $0.2$ ). The model with FINNpeatSM emissions has the smallest bias (average FB =  $-0.11$ ), suggesting mean emissions from this inventory are closest to reality.

The temporal pattern of measured PM is generally matched by the simulations, as shown in Fig. 4. However, for many sites, the greatest PM concentrations were measured in October, whereas the model simulates the greatest PM concentrations in September. This results in the model underestimating PM concentrations the most in October, with a smaller underestimate, or an overestimate, in September (Fig. 5b).

Using a burn depth dependant on soil moisture alters the temporal pattern of simulated emissions, reducing the overestimation in September compared to October. When burn depth is constant, as in FINNpeat, 37 % of regional PM<sub>2.5</sub> emissions for 2015 occur in September and 30 % in Octo-



**Figure 4.** Daily observed and modelled (a)  $\text{PM}_{2.5}$  in Singapore and (b)  $\text{PM}_{10}$  in Pekanbaru for WRF-chem runs using different fire emissions inventories and the surface injection option. Panel (a) shows observations of  $\text{PM}_{2.5}$  (solid) and  $\text{PM}_{10}$  (dashed). The Pearson's correlation ( $r$ ) for panel (a) is 0.47, 0.73, 0.52 and 0.50 and for panel (b) is 0.63, 0.60, 0.65 and 0.73 for FINN, FINN+GFEDpeat, FINNpeat and FINNpeatSM, respectively.

ber. In FINNpeatSM, where we assume a linear relationship between soil moisture and burn depth, the percentage of annual  $\text{PM}_{2.5}$  emissions in September is 39 % and 36 % in October. A non-linear relationship between soil moisture and burn depth would result in shallower burn depth in September and deeper burn depth in October, decreasing emissions in September and increasing emissions in October, which might further improve simulated PM concentrations. There is little information available on the measured relationship between soil moisture and burn depth.

The overestimation of modelled PM concentrations in September may also be due to our assumption that all the emissions from a fire are emitted on the day the fire was detected. In reality, peat fires can smoulder for weeks and the emissions should be released over a longer time period. This could also reduce the simulated PM concentration in September and increase them in October. The overestimation in September could also be due to an issue with fire detection. Syaufina and Sitanggang (2018) found that only hotspots which last for at least 3 consecutive days indicate fires, something which is not considered when calculating the emissions. However, despite our simplified assumptions

the model captures individual peaks in measured PM reasonably well (Fig. 4). Alternately, an underestimation in October could be due to clouds, or haze caused by previous fires, blocking the detection of fires by the satellite.

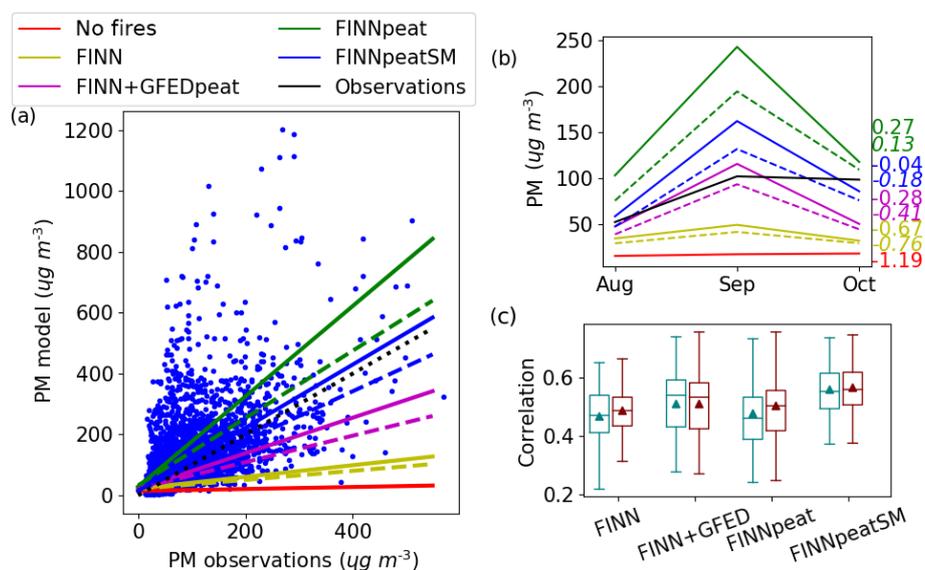
Putriningrum et al. (2017) found that WRF-chem with FINNv1 or GFEDv4 emissions underestimated PM concentrations across Indonesia during October 2015, with GFEDv4 resulting in a better match compared to FINNv1. Putriningrum et al. (2017), suggested that emissions were underestimated due to haze from fires blocking the detection of burned area. While this is likely to affect the emissions, our work suggests that PM emissions in GFED4 could also be underestimated because EFs for peat combustion are too small.

Figure 5 also shows the correlation coefficients between modelled and measured PM concentrations across all the observation sites. The FINN simulation has the lowest average correlation across all sites ( $r = 0.47$  and  $0.49$  with surface and boundary layer injection, respectively), followed by FINNpeat ( $r = 0.48$  and  $0.51$ ) and FINN+GFEDpeat ( $r = 0.51$  for both injections). FINNpeatSM has the highest average correlation across all the sites ( $r = 0.56$  to  $0.57$ ). Both FINN+GFEDpeat and FINNpeatSM assume variable peat burn depth depending on soil moisture. This comparison therefore suggests that varying depth of peat burn based on surface soil moisture, as in FINNpeatSM and FINN+GFEDpeat, results in improved estimate of emissions. The height at which emissions are injected has little impact on the correlation, so there is limited evidence from this comparison to support either option.

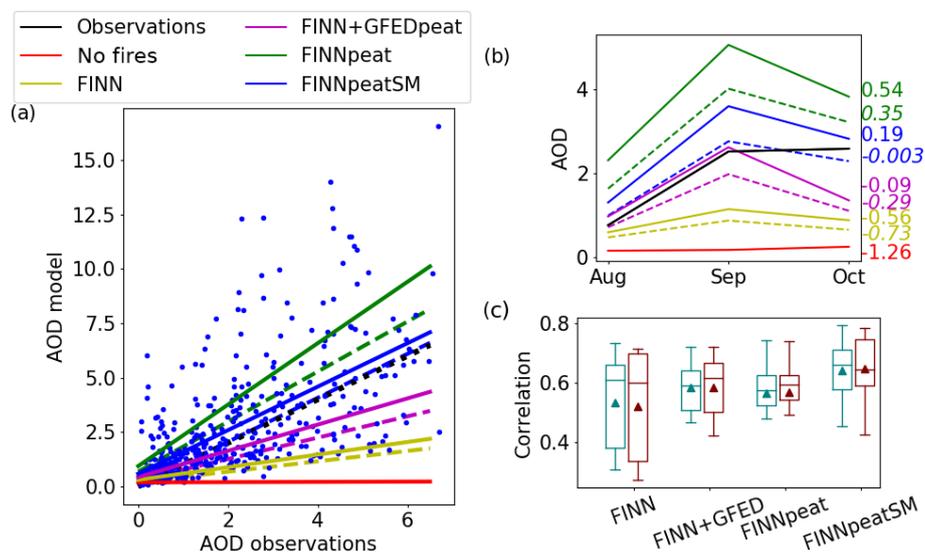
Comparison with  $\text{PM}_{2.5}$  concentrations measured in Singapore during October 2015 further supports the above analysis. WRF-chem underestimates  $\text{PM}_{2.5}$  concentrations in Singapore both with FINN emissions (FB =  $-0.6$  for surface emissions and  $-0.69$  for boundary layer emissions) and FINN+GFED emissions (FB =  $-0.28$  for both injections). With FINNpeat emissions the model overestimates  $\text{PM}_{2.5}$  concentrations (FB =  $0.45$  to  $0.53$ ) and the best agreement with observations is with FINNpeatSM emissions (FB =  $0.06$  to  $0.16$ ).

Chemically resolved  $\text{PM}_{10}$  measurements from Singapore are available for 10–31 October 2015. Organic aerosols (OAs) contributed 79 % of the observed  $\text{PM}_{10}$  between 10 and 31 October (Budisulistiorini et al., 2018). The FINN simulation underestimates the contribution of OA to  $\text{PM}_{10}$  with 64 % with BL injection (69 % with surface injection). For the simulations with peat emissions, the model is improved with the contribution of OA to  $\text{PM}_{10}$  varying (Fig. S4). With FINNpeatSM, 78 % of  $\text{PM}_{10}$  is OA with the boundary layer injection (82 %, surface injection). For the simulations with FINNpeat it is 80 % (84 %), for FINN+GFED 78 % (79 %).

Figure 6 shows comparison of simulated and measured AOD. The comparisons are consistent with those seen for PM. The model with FINN emissions underestimates AOD (FB =  $-0.56$  for surface emissions and  $-0.73$  for bound-



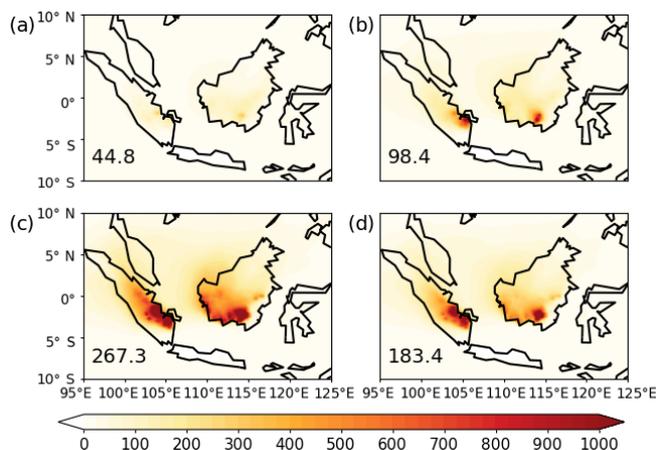
**Figure 5.** Comparison of simulated and observed PM concentrations during August–October 2015. Observations of  $\text{PM}_{10}$ ,  $\text{PM}_{2.5}$  and  $\text{PM}_1$  from 55 sites in Indonesia, Singapore and Malaysia. **(a)** Simulated and observed daily mean PM concentrations for FINNpeatSM emissions and surface injection (blue dots). Lines show the linear fit for the model with different emissions: solid lines are when emissions are emitted at the surface and dashed lines are used when emissions are injected into the boundary layer. The 1 : 1 line is shown with black dots. **(b)** The average monthly simulated and observed PM concentrations. The fractional bias for August–October is shown to the right of each line. **(c)** The correlation coefficient ( $r$ ) for comparisons of daily mean simulated and observed PM concentrations for all 55 sites. For each simulation the box plots show the median (middle line of box) and upper and lower quartiles (top and bottom of box). The range of correlations (whiskers extend to min and max) across all sites are shown by the box plots, and the mean correlations are shown as triangles. Simulations with the surface injection are in light blue, and simulations with the boundary layer injection are in red.



**Figure 6.** Comparison of simulated and observed AOD during August–October 2015, from eight AERONET sites in Indonesia, Singapore and Malaysia. Observed AOD is at 500 nm and simulated AOD is at 550 nm. Panels **(a)**, **(b)** and **(c)** are the same as in Fig. 5 but for AOD.

ary layer), as does the model with FINN+GFED emissions (FB =  $-0.09$  and  $-0.29$ ). FINNpeat overestimates for both injection options (FB =  $0.54$  and  $0.35$ ), and FINNpeatSM gives the lowest FB of  $-0.003$  with boundary layer injection ( $0.19$  with surface injection). The correlation coefficients

between simulated and measured AOD are highest for simulations with FINNpeatSM ( $r = 0.64$  with surface and  $0.65$  with BL injection) followed by FINN+GFEDpeat ( $r = 0.58$  and  $0.59$ ), FINNpeat ( $r = 0.57$  for both injections) and FINN ( $r = 0.53$  and  $0.52$ ). The AOD simulated by the model ex-



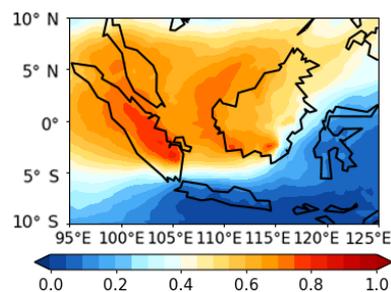
**Figure 7.** Mean simulated surface  $\text{PM}_{2.5}$  concentration ( $\mu\text{g m}^{-3}$ ) from fires for September–October 2015 with the surface injection and (a) FINN emissions, (b) FINN+GFEDpeat, (c) FINNpeat and (d) FINNpeatSM. The surface  $\text{PM}_{2.5}$  concentration from fires, averaged over Borneo and Sumatra, is indicated on each panel.

ceeded 10 during September–October, similar to the values estimated by Eck et al. (2019) for the same period. Previous work has found that models tend to better simulate  $\text{PM}_{2.5}$  compared to AOD in regions influenced by fire emissions (Aouizerats et al., 2015; Crippa et al., 2016).

### 3.3 $\text{PM}_{2.5}$ concentrations and AOD

Figure 7 shows simulated surface  $\text{PM}_{2.5}$  concentrations due to fires during September–October (Fig. S5 shows results for the boundary layer injection). Simulated  $\text{PM}_{2.5}$  concentrations from fires are greatest over Sumatra and southern Kalimantan, with simulated September–October mean concentrations exceeding  $1800 \mu\text{g m}^{-3}$  in some grid cells in FINNpeatSM emissions. Enhanced regional  $\text{PM}_{2.5}$  concentrations are simulated to the northeast of the fires across Peninsular Malaysia ( $50$ – $150 \mu\text{g m}^{-3}$ ) caused by regional transport of pollution. Simulated surface  $\text{PM}_{2.5}$  concentrations from fires during September–October over Sumatra and Borneo are greatest with FINNpeat emissions ( $267 \mu\text{g m}^{-3}$ ), followed by FINNpeatSM ( $183 \mu\text{g m}^{-3}$ ), FINN+GFEDpeat ( $98 \mu\text{g m}^{-3}$ ) and FINN ( $45 \mu\text{g m}^{-3}$ ), matching the  $\text{PM}_{2.5}$  emissions from the different inventories (Table 4).

Peat combustion contributes a substantial fraction of simulated  $\text{PM}_{2.5}$  concentrations from fires, ranging from 55 % in the model with FINN+GFEDpeat emissions, to 76 % with FINNpeatSM emissions and to 83 % with FINNpeat emissions. Figure 8 shows the fraction of the simulated surface  $\text{PM}_{2.5}$  concentration from peat fires for September–October 2015 using the FINNpeatSM emissions. The majority of simulated  $\text{PM}_{2.5}$  concentrations across the study area are due to emissions from peat fires. Across Sumatra and Borneo, 96 % of surface  $\text{PM}_{2.5}$  concentrations are from fires with 73 % from peat combustion. Peat fires therefore account for

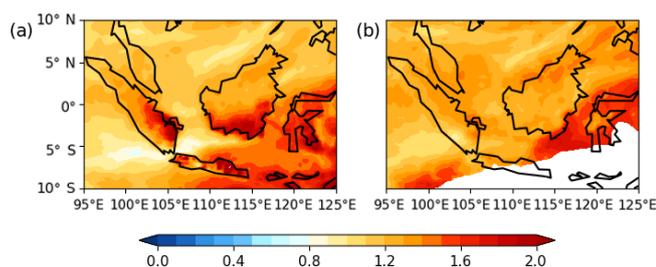


**Figure 8.** Fraction of simulated  $\text{PM}_{2.5}$  concentrations originating from peat fire emissions. Simulations use the new FINNpeatSM fire emissions with surface injection.

76 % of the fire contribution to  $\text{PM}_{2.5}$ . This is slightly larger than the contribution of peat fires to primary  $\text{PM}_{2.5}$  emissions (71 % in FINNpeatSM), likely due to atmospheric production of secondary organic aerosol from fire-emitted precursors. Reddington et al. (2014) used a combination of models to demonstrate that regional fire-derived PM concentrations during haze episodes are dominated by emissions from peatland regions. For 2015, Wiggins et al. (2018) suggest that  $\sim 85$  % of the smoke reaching Singapore was from peat fires, slightly higher than the contribution of peat fires to the simulated  $\text{PM}_{2.5}$  concentration shown in Fig. 8 (67 %). Engling et al. (2014) found that, in 2006, 76 % of particulate matter in Singapore was from peat fires. At the Palangkaraya AERONET site in Kalimantan, Eck et al. (2019) found that 80 %–85 % of AOD came from peat burning, consistent with the contribution of peat fires to  $\text{PM}_{2.5}$  simulated by the model at that location (Fig. 8).

Inclusion of emissions from peat fires gives the largest fire emissions in Sumatra and western Kalimantan, where previously emissions were substantially lower (Fig. 2). This leads to higher  $\text{PM}_{2.5}$  concentrations across Singapore (Fig. 5), which has a large impact on the population exposure to the  $\text{PM}_{2.5}$ . The location of fires can be an important factor of their contribution to air pollution in populated areas. Lee et al. (2017) found that the 2015 fires in Sumatra accounted for 50 % of fire-derived  $\text{PM}_{2.5}$  in Kuala Lumpur and 41 % in Singapore, and Hansen et al. (2019) found that during August–October fires in southern Sumatra and central Kalimantan are the largest contributors to  $\text{PM}_{2.5}$  in Singapore. Reddington et al. (2014) and Kim et al. (2015) found that for the 2006 fires Sumatran fires were responsible for the worst air quality across Equatorial Asia.

Injecting all fire emissions at the surface increases the average simulated surface  $\text{PM}_{2.5}$  concentration by a factor of 1.34 to 1.36, compared to injecting 50 % at the surface and 50 % through the boundary layer. However, this factor varies spatially (Fig. 9). Close to the fire locations, the surface injection option results in an increase in  $\text{PM}_{2.5}$  concentrations by up to a factor of 2. Further away from the fires, however, the injection option has less impact on simulated  $\text{PM}_{2.5}$  con-



**Figure 9.** Ratio of simulated (a) surface  $\text{PM}_{2.5}$  concentration and (b) AOD at 550 nm from fires for September–October, when using the surface injection option compared to the boundary layer injection option. Results are shown for the model with FINNpeatSM emissions. Zero values of average  $\text{PM}_{2.5}$  and AOD have been removed.

centrations. Despite these differences in simulated PM, the available measurements of PM do not allow us to better constrain the vertical profile of fire emissions (Sect. 3.2).

Simulated AOD in Sumatra and Borneo during September–October (Fig. S6) follows a similar pattern to simulated  $\text{PM}_{2.5}$ , with the highest value for the model with FINNpeat (1.42) followed by FINNpeatSM (1.06), FINN+GFEDpeat (0.62) and FINN (0.31). Injecting fire emissions at the surface also results in greater simulated AOD compared to when emissions are spread through the boundary layer. The average AOD across Borneo and Sumatra increases by a factor of 1.32 for surface-injected emissions compared to the boundary layer, which is similar to the difference seen in average  $\text{PM}_{2.5}$  concentrations. However, spatially the difference between the injection options is different from that seen for  $\text{PM}_{2.5}$  (Fig. 8). Rather than seeing a larger increase around the fires in Sumatra and Kalimantan, the factor difference between the two injection options remains between 1.1 and 1.5 across the area affected by fires. Majdi et al. (2019) also found that the sensitivity of simulated surface  $\text{PM}_{2.5}$  to injection method (up to 50 %), was greater than the sensitivity of AOD (up to 20 %), which is consistent with the differences seen here close to fires.

#### 4 Conclusions

Vegetation and peat fires in Indonesia emit substantial amounts of trace gases and aerosol resulting in serious air pollution episodes. The magnitude of emissions from these fires is very uncertain, particularly for peat fires, which are more difficult to detect using Earth observation methods. New measurements of tropical peat combustion have led to an upward revision of particulate emission factors, leading to a suggestion that some fire emission inventories may underestimate particulate emissions from peat fires. Here we used the WRF-chem model along with extensive observations of PM to make a revised estimate of PM emissions from Indonesian fires during August–October 2015.

Current fire emission inventories either do not include peat fires, (FINNv1.5), or do not use updated peat emission factors (GFEDv4s). The WRF-chem model underestimated PM concentrations measured in Indonesia and Malaysia during August–October 2015, both with FINNv1.5 emissions (fractional bias =  $-0.7$ ) and with a combination of FINN vegetation emissions and GFED4s peat emissions (fractional bias =  $-0.35$ ). We created a new emissions inventory for Indonesia using updated emission factors for peat combustion and with variable assumptions relating the depth of peat burn to soil moisture (FINNpeatSM). Our best emissions estimate, FINNpeatSM, leads to an improved simulation of PM concentrations (fractional bias =  $-0.11$ ). Estimated  $\text{PM}_{2.5}$  emissions from fires across Sumatra and Borneo during September–October 2015 are 7.33 Tg (with FINNpeatSM), a factor 1.8 greater than in GFED4 (4.14 Tg) and a factor 3.5 greater than FINN1.5 (2.1 Tg). Our total emissions agree with estimations by Wooster et al. (2018) ( $9.1 \pm 3.2$ ) and Jayaranthe et al. (2018) ( $6 \pm 5.5$  Tg from peat fires). Further work is needed to assess the impacts of secondary processes within the model on  $\text{PM}_{2.5}$  concentrations, and how this may affect the comparisons between model and observations made in this study. We estimate that fires contributed an additional 3.12 Tg of secondary organic aerosol emissions, equivalent to 31 % of total emissions from fires. This brings the total PM from fire emissions to 10.45 Tg. Since updated  $\text{CO}_2$  EFs for peat fires are similar to previous measurements, our estimated  $\text{CO}_2$  emissions are consistent with GFED4s.

We find that emissions from peat combustion make up a substantial fraction of total fire emissions from the region. We estimate that peat combustion contributed 55 % of total  $\text{CO}_2$  emissions and 71 % of primary  $\text{PM}_{2.5}$  emissions during September–October 2015. Peat combustion contributed 76 % of fire-derived surface  $\text{PM}_{2.5}$  concentrations over Sumatra and Borneo during this period. This highlights the importance of peat fires and the need for better estimates of emissions from peat combustion.

The depth of peat burn is a crucial factor controlling emissions from peat fires but it is poorly constrained. We found that using satellite remotely sensed soil moisture to control the assumed depth of peat burn improved the simulation of PM, with the correlation between simulated and measured PM increasing from 0.48 with fixed peat burn depth to 0.56 with soil moisture control. There is little data available on the relationship between surface soil moisture and burn depth, more work on this could lead to further improvement in the simulation. Work is also needed to examine whether this is consistent for years other than 2015.

Our work suggests that existing emission inventories (GFED4 and FINN1.5) underestimate particulate emissions from Indonesian fires, due to an underestimation of particulate emissions from peat combustion. Including updated emission factors from tropical peat combustion results in substantially increased PM emissions from Indonesian fires. Measurements of emission factors from tropical peat com-

bustion are still very limited, and therefore additional measurements are required. Our comparison of simulated and measured PM concentrations across the region provides an additional and independent confirmation of updated emission factors from peat combustion. Our work suggests that previous studies may have underestimated the contribution of Indonesian fires on particulate air quality. We estimate that vegetation and peat fires increased PM<sub>2,5</sub> concentrations over Sumatra and Borneo during September–October 2015 by an average of 127 µg m<sup>-3</sup>. Future work needs to explore the impact of these fires on public health.

*Code and data availability.* Code and data used in this study are available from authors upon request.

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*Author contributions.* Conceptualization by LK and DVS, with support from SRA and LS. LK carried out modelling work and analysis, with help from DVS and extra supervision from SRA. CK provided WRFotron modelling scripts, and SAN and DL provided model code and technical support with modelling. LC and CLR provided technical support with modelling. LK and CW provided fire emissions, and MK, SHB, MFK and MTL all provided observations. LK prepared the manuscript with help from all authors.

*Competing interests.* The authors declare that they have no conflict of interest.

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