Identifying and addressing knowledge gaps for improving greenhouse gas emissions estimates from tropical peat forest fires

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Abstract

Tropical peatlands are areas of high carbon density that are important in biosphere-atmosphere interactions. Drainage and burning of tropical peatlands releases about five percent of global greenhouse gas (GHG) emissions, yet there is great uncertainty in these estimates. Our comprehensive literature review of parameters required to calculate GHG emissions from burnt peat forests, following the international guidelines revealed many gaps in knowledge of carbon pools and few recent supporting studies. To improve future estimates of the total ecosystem carbon balance and peatfire emissions this study aimed to account for all carbon pools: aboveground, deadwood, pyrogenic carbon (PyC) and peat of single and repeatedly burnt peat forests. A further aim was to identify the minimum sampling intensity required to detect with 80% power significant differences in these carbon pools among long unburnt, recently burnt and repeatedly burnt peat swamp forests.

About 90 Mg C ha^{-1} remains aboveground as deadwood after a single fire and half of this remains after a second fire. One fire produces 4.5±0.6 Mg C ha^{-1} of PyC, with a second fire increasing this to 7.1±0.8 Mg C ha^{-1}. For peat swamp forests these aboveground carbon pools are rarely accounted in estimates of emissions following multiple fires, while PyC has not been included in the total peat carbon mass balance. Peat bulk density and peat carbon content change with fire frequency, yet these parameters often remain constant in the published emission estimates following a single and multiple fires. Our power analysis indicated that as few as 12 plots are required to detect meaningful differences between fire treatments for the major carbon pools. Further field studies directed at improving the parameters for calculating carbon balance of disturbed peat forest ecosystems are required to better constrain peatfire GHG emission estimates.

Keywords: biomass burning, Indonesia, repeated fires, pyrogenic carbon, deadwood, power analysis

1. Introduction

Tropical peatlands are areas of high carbon density that play an important role in biosphere-atmosphere interactions (Canadell et al., 2004). They cover about 182 million ha across South America, Africa and Asia; the latter accounts for about 20.3% (36.9 million ha) of the total area (Leng et al., 2019). Within Asia, Indonesia has the largest area (20.7 million ha) and the largest share of tropical peat carbon (57.4 Gt, 65%) of the global total for peatlands (Page et al., 2011). Rapid degradation of peatlands continues around the globe, including in Indonesia, where they are converted to agriculture and plantations for palm oil and wood pulp, and subject to timber extraction (Koh et al., 2009). Drainage of peatlands for agriculture and plantations dries out surface peat over extensive areas, making them susceptible to recurring fires, especially during recent extended dry spells associated with global warming. Repeated and extensive fires, following drainage and selective logging, played an important role in peat forest loss in Indonesia over 1973-2005 (Hoscilo et al., 2011). Regional droughts in 1997-98, 2005, 2015-16, and 2019 resulted in an unprecedented increase in peat fires in Indonesia, affecting both natural forests and those subjected to conversion to plantations. Smoke and air pollution from those fires affected not only Indonesia but all countries in Southeast Asia (Hayasaka et al., 2014; Marlier et al., 2015; Tham et al., 2019; Wiggins et al., 2018). Reducing smoke and emissions from peat fires is important for health and air quality, it is also gaining national and international significance as a mechanism for addressing climate change (UNFCCC, 2015). For these reasons the Food and Agriculture Organisation (FAO) of the United Nations (UN) has recently declared that improving the assessment of greenhouse gas emissions (GHG) from peatland is a global strategic priority (FAO and Wetlands International, 2012).

Following the Intergovernmental Panel on Climate Change (IPCC) guidelines, GHG emissions from peat fires are estimated as the sum of emissions from burning of aboveground (AG) carbon stocks and combustion of peat, using Equation 2.27 of the IPCC (2006a) and Equation 2.8 of the IPCC (2014), (Eq 1):

$$E_i = A \cdot [(M_{AG} \cdot C_{AG} \cdot CF_{AG} \cdot Gef_{i\_AG}) + (M_{PEAT} \cdot C_{PEAT} \cdot CF_{PEAT} \cdot Gef_{i\_PEAT})] \cdot 10^{-3}$$

(Eq.1)

Where: E_i is emission for the i^{th} direct or indirect GHG; the i^{th} direct GHGs are CO_2, CH_4, N_2O, and indirect GHGs are CO, NO_x and VOC (volatile organic compounds), Gg; A is the peat burnt area, ha; M_{AG} is mass of aboveground fuel (live biomass, litter and deadwood) available for combustion, Mg ha^{-1}; C_{AG} is a carbon mass fraction in aboveground fuels which is required for all carbon and nitrogen emissions estimates. CF_{AG} is the combustion factor of aboveground fuels (AG) estimated as the difference in the aboveground fuel before and after fire, unitless; and, Gef_{i_AG} is a gas-specific emission factor or the amount of the i^{th} GHG released per kg of dry AG matter burnt, g kg^{-1}. M_{PEAT} is the mass of dry peat, Mg ha^{-1}. M_{PEAT} is calculated from peat bulk density (BD), g cm^{-3} multiplied by peat depth loss h, cm. C_{PEAT} is peat carbon concentration, required for all carbon and nitrogen emissions estimates. CF_{PEAT} is peat combustion factor; and, Gef_{i_PEAT} is gas-specific emission factor or the amount of the i^{th} GHG released per kg of dry peat burnt, g kg^{-1}.

If there is a lack of country-specific data to estimate peat fire emissions, the IPCC provides default parameters for M, C, CF and Gef_i based on a limited number of studies; these are shown in Tables 2.4-2.6 of IPCC (2006a), and in Tables 2.6-2.7 of IPCC (2014). For tropical peatland the default CF_{AG} is 0.50 (based on one study of Levine 2000) and CF_{PEAT} is 1.0 (or 100% combustion).

With this background in mind our study firstly reviews the supporting data in the literature to identify knowledge gaps for improving GHG emissions estimates. Secondly, with a view to improve emission parameters, a field study is then applied to determine the sampling intensity to identify differences in emission parameters among a range of peat forests burnt at different fire frequencies.

2. Knowledge gaps in the emissions estimates

2.1 Knowledge gaps in the emissions parameters

Although emissions from tropical peat forest fires have been the subject of hundreds of publications[1], most studies do not improve knowledge of parameters required for the IPCC emissions equations. Current emissions from drained or burnt Indonesian peatlands are claimed to be in the range of 2 Billion t CO_2 per year, accounting for about 5% of the global carbon budget (UN, 2017), yet there is limited transparency in these estimates. Because there are very few studies to support country-specific conditions, the IPCC default parameters are often used. The IPCC default parameters for such a significant carbon pool as M_{PEAT} are derived from just three studies (Ballhorn et al., 2009; Page et al., 2002; Usup et al., 2004). Emissions from aboveground fuels are often excluded; for example, among published estimates of Indonesian peat fire emissions, we found only a few original studies that measured (and included) losses from aboveground dead biomass (e.g. Siahaan et al., 2020; Toriyama et al., 2014). Furthermore, emission estimates derived from complex biogeochemical models lack empirical data to reduce uncertainty in predictions. For example, the recently released Global Carbon Budget (Friedlingstein et al., 2019) estimates peat fire emissions using a complex model (the Global Fire Emission Database, GFED4s), yet it provides similar estimates for Indonesian peat fire emissions to estimates made with default parameters of the IPCC (Prosperi et al., 2020; Rossi et al., 2016).

In its first Nationally Determined Contribution (NDC), Indonesia committed to reduce its GHG emissions by 26% relative to a business as usual scenario by 2020, and by 41% with international support. For the period from 2020 to 2030 these reduction targets are 29% (unconditional) and 41% (conditional) (Republic of Indonesia, 2016). Due to a high level of uncertainty in peat fire emissions parameters, the Indonesian Government excluded emissions from peat fires in its first Forest

[1] Above 500,000 results from Google search for 'Peat fire emissions AND Indonesia' and around a thousand from the Web of Science using a combination of words' peat fire, carbon loss, tropical' at 20 April of 2020.

Reference Emission Level (FREL) submitted to the UNFCCC (MoEF, 2016), preventing it from subsequently claiming emission reduction from reduced peat fires.

There are not many studies which support fire emission parameters of Eq 1 in a comprehensive way. Moreover, the majority of studies report field data collected at least ten or more years ago (Table 1). With few recent empirical studies on which to improve emission estimates the IPCC was unable to update emissions parameters in the 2006 Guidelines for the 2019 refinement (IPCC, 2019).

Table 1 Review of the parameters contributing to tropical peat fire emissions estimates as per IPCC Guidelines, for the detail of studies refer to Supplementary Table S1

Study	M_{AG}	CF_{AG}	M_{PEAT}		C_{PEAT}	CF_{PEAT}	Gef_{i_PEAT}	***Date collected***
			h	BD				
1					x			2013-2014
2				x	x			2007-2010
3	x	*	x	x				2014-2015
4			x					2007
5			x					1997
6							x	n/g
7	1	*						2014
8	2							2011
9			x	x				2011
10							x	2009
11			x	x	x			2010-2013
12							x	2015
13							x	2003
14	x	*			x			2005
15							x	2015
16			x	x	x			2010-2011
17				x				2000
18	1	*						2007-2008
19	x	*						2001
20							x	2009
21				x	x			1969-2012
22			x	x	x			1999/2000
23				x	x			1995
24	x	*		x	x			2009
25				x	x		x	2016
26	3	x						2001-2002
27	1			x	x			2015

28				x	x			1997-1999
29	x	*						2015
30			x					2015
31				x				2013
32				x	x		x	2016
33							x	n/g
34			x				x	2015
35	x	*						2009
36	x	x	x			x		2002
37				x	x			n/g
38				x	x		x	2015

* can be extrapolated from the data; n/g – not given; 1– trees only; 2 – only volume of CWD; 3 – litter and branches only.

In recognizing the general lack of new or recent literature to support emission estimates, we provide here a detailed overview of the parameters required to estimate peat fire emissions as a commentary to information presented in Table 1:

$M_{AG} \cdot CF_{AG}$ – the mass of aboveground fuel and its combustion factor are covered by fewer than 10 studies, where only three cover all aboveground fuels, and a further two studies that measured CF_{AG} as pre- to post-fire mass difference. Fewer than five studies report losses of aboveground fuels resulting from multiple fires. There seems to be no common or standardised approach for measuring and reporting all aboveground fuels in peat forests.

M_{PEAT} – there are fewer than ten studies providing information on the critical peat loss (burn depth) parameter. Only Usup et al. (2004) reports on belowground fuels such as grass roots and submerged woody debris. While there is a reasonable coverage in the literature on peat *BD* (Table 1), and also see the often referenced studies (e.g. Neuzil, 1997; Supardi et al., 1993); only a few studies report changes in peat *BD* following one or more fires (e.g. Konecny et al., 2016; Sinclair et al., 2020). Although there are numerous studies of C_{PEAT}, none report the impact of fire or frequent fires on this parameter. Given strong evidence for the fire-modification of soil organic carbon, this is a major knowledge gap given the extent of frequently burnt peatlands.

CF_{PEAT} – a value of 1 is universally applied and assumes complete combustion of peat (Usup et al., 2004); it is acknowledged in the literature as an oversimplification (Hooijer et al., 2014; Konecny et al., 2016).

Gef$_{iAG}$ parameter, not specific to peat swamp forests, has a good coverage in the literature from studies for other forests types (Andreae, 2019). *Gef*$_{iPEAT}$ is a reasonably well-studied parameter in the laboratory and in the field; also see reviews (Hu et al., 2018; Levine, 2000).

Based on this evaluation of the current literature, we suggest that to improve emission estimates from tropical peatlands, attention should first be focused on the aboveground fuels ($M_{AG} \cdot CF_{AG}$) and peat ($h \cdot BD \cdot C_{PEAT}$), as these are major determinants of estimates.

2.2.Knowledge gaps of the effect of frequent fires on emissions parameters

Over the last two decades about 12% of peatlands in Sumatra and Kalimantan have been burnt more than once, with about 23% of this area burnt more than twice (Vetrita and Cochrane, 2020). Results from just a few studies of degraded and repeatedly burnt forests reveal about 80% loss of aboveground biomass, with litter (5 Mg C ha^{-1}) and deadwood (26 Mg C ha^{-1}) accounting for most of that total (Dharmawan, 2012; Qirom et al., 2018). There is an urgent need for improved data on aboveground carbon stocks and the mass of fuel burnt, so that uncertainty in peat fire emission estimates can be reduced (Austin et al., 2018).

2.3 Knowledge gaps on the importance of pyrogenic carbon for the emissions estimates

Current peat emission estimates do not account for the production of PyC, the thermochemically altered biomass that is created from the pyrolysis and incomplete combustion of organic matter (also referred to as black carbon, soot, elemental carbon, char, charcoal and biochar) (Bird et al., 2015; Surawski et al., 2020). Globally, vegetation fires can produce about 116-385 Tg PyC per year (Santin et al., 2016). In cold temperate peatlands between 2% to 4% of combusted biomass is converted to PyC in fires (Worrall et al., 2013); there are no corresponding studies of PyC production following Indonesian peat fires. Accounting for PyC will assist in constraining carbon loss estimates and improving the accuracy of emissions estimates. Ignoring production of PyC, assumes that all biomass consumed in fires is emitted, globally leading to the annual over estimation of carbon emission by 100 Tg (Surawski et al., 2016). Furthermore, PyC that is produced from biomass burning and not emitted to the atmosphere is a potential source of long-term carbon sequestration when stored in soils or sediments (Preston and Schmidt, 2006). Char has been shown to have mean residence times of up to 10000 years in soils (Swift, 2001); this relative inertness means that it must be considered as a significant component of global cycling (Forbes et al., 2006). Consequently, accounting for PyC in the carbon mass balance will further help to improve the accuracy of peat forest fire emission estimates.

3. Addressing knowledge gaps in the emissions parameters

To address the gaps in emission parameters identified above, a feasible field sampling design, with sufficient replication of biomass components (aboveground live, deadwood, litter, PyC and peat carbon) is required to understand the contribution of individual carbon pools to total peat forest carbon. The sampling intensity for biomass components should enable detection of significant differences in the carbon pools between fire treatments.

For researchers studying the impact of fire regimes on carbon balance of peat forests, sampling design and sample size become non-trivial questions. Long undisturbed peat forests have a dense canopy of overstorey trees that account for the majority of aboveground carbon (Verwer and van der Meer, 2018), while recently and repeatedly burnt peat forest is either dominated by vigorous young regrowth or dominated by bare ground, with a low density of dead trees remaining from previous fires (Konecny et al., 2016; Siahaan et al., 2020). Sampling designs must be flexible enough to characterise differences in the allocation of carbon among pools in relation to the number of fires and fire return interval. Fixed plot designs, such as circular plots, are usually straight-forward to establish in most forest settings, and are traditional choices in many forest inventories (McRoberts et al., 2013). Yet, when site conditions are variable it is often unclear how many samples are required to test the hypotheses of interest. This problem is likely exacerbated when collecting data from systems exposed to novel disturbances, as the variance associated with the data is unknown.

It is a good practice to conduct a pilot study, to design an experiment resulting in analyses with sufficient statistical power to address the research question. Statistical power is defined as the probability of correctly detecting a significant effect if it exists in the population of interest; mathematically $Power = 1 - \beta$ where β is probability of making a Type II error – failing to detect an effect if it exists. The power of an analysis is related to sample size and variance, but also to the effect size (e.g., magnitude of the difference between two groups: a treatment and control) that researchers deem important to detect (Di Stefano, 2001; Foster, 2001). Generally, the smaller the effect size, the greater the sample size required. Researchers must decide the meaningful magnitude of change that is important (Westfall et al., 2013). Chasing a small effect size may also lead to over-spending resources or to finding an effect that is either small or doesn't really exist (Hoenig and Heisey, 2001). Statistical power analysis can help to find the minimum sampling effort needed to detect relevant differences between treatments and changes over time in monitored variables (Di Stefano, 2001; Foster, 2001).

In recognizing a general lack of supporting data in the literature, we aimed to address knowledge gaps identified in Table 1 through a field study where we first deal with the sampling intensity required to achieve sufficient power for our research purpose of identifying treatment differences. Specifically, the main aims of this study were: i) to develop practical and achievable field sampling designs for aboveground fuels and peat components in repeatedly burnt peat swamp forests; ii) to identify the impact of recent and repeated fires on peat swamp forest carbon pools and, iii) to identify sampling intensity required to detect with 80% power significant differences in biomass carbon pools (e.g., aboveground live, deadwood) of long unburnt, recently burnt and repeatedly burnt peat swamp forests.

4. Material and Methods

4.1 Study sites

An area of degraded peat swamp forest of the former Mega Rice project of Central Kalimantan, Borneo, was selected for the study (Fig.1). Specifically, the plots were located within Tumbang Nusa Research Forest (KHDTK Tumbang Nusa) (0°8'48" to 3°27'00" South and 113°2'36" to 114°44'00" East), about 30 km south east of Palangka Raya, Indonesia. Throughout 2019 and early 2020 a pilot study of 18 plots comprised of 2 blocks x 3 treatments x 3 plots was sampled. Plots were selected to cover different fire history with accessibility in the difficult terrain a key factor due to time constraints and limited resources. Once the treatment locations were identified, selection of the first plot was random, followed by a grid method with a minimum distance of 60 m between plots along a pre-determined compass bearing to locate the second and subsequent plots.

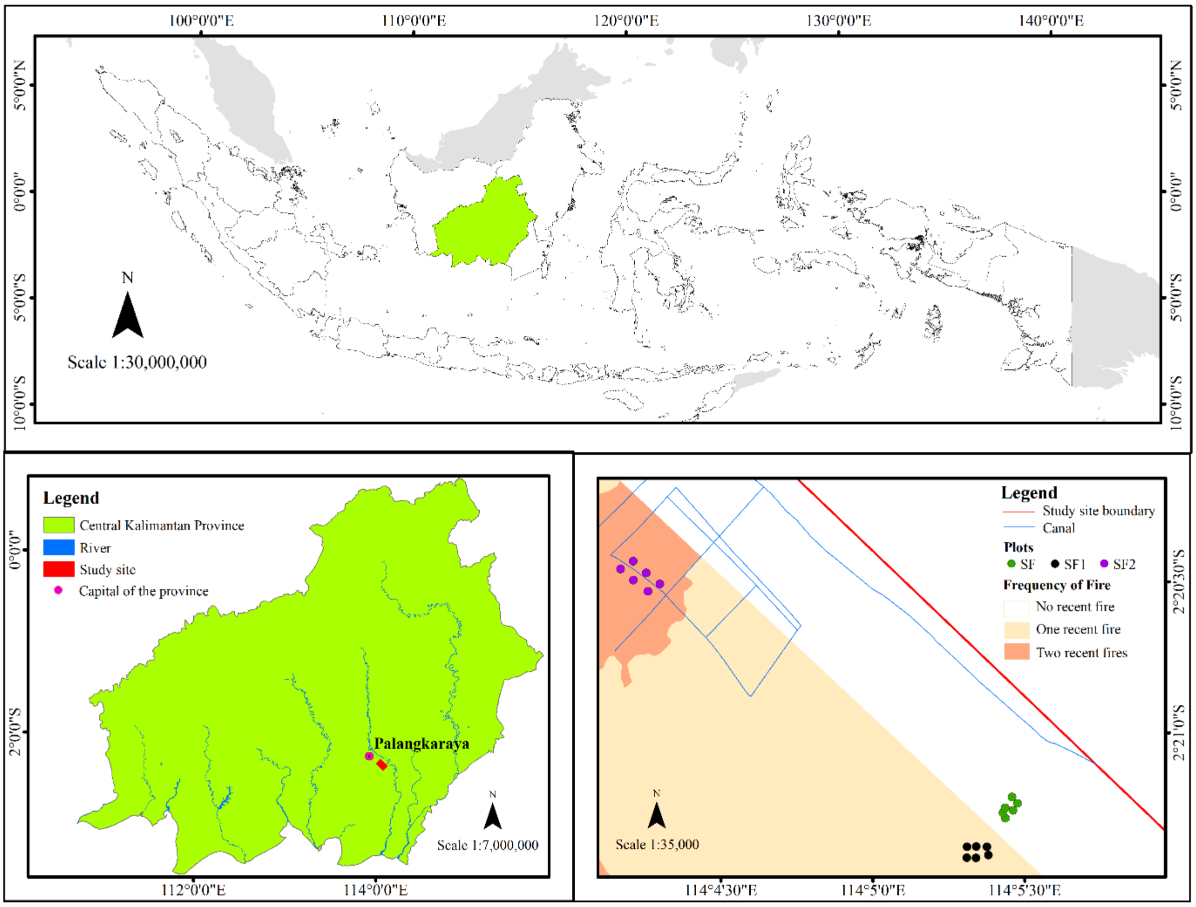

Figure 1. Study sites location in Central Kalimantan, Indonesia

4.2 Treatments

An extensive fire in 1997 burnt through all the Tumbang Nusa area of peat swamp forest that had been drained and logged throughout the 1980s and 1990s. Following the 1997 fire some areas of forest regenerated without subsequent burning while other areas burnt again in 2014 and 2015 (fire history data were provided by the Banjarbaru Forestry and Environment R&D Institute). For this study the following treatments were selected: SF – Secondary peat swamp forest regenerated though natural processes, long unburnt (fire in 1997); SF_1 – Secondary peat swamp forest burnt in one recent fire – a fire in 2015 and also in 1997, and SF_2 – Secondary peat swamp forests burnt in two recent fires - fires in 2015 and 2014 and also in 1997, (Figs. 1-2).

Given the different structure of aboveground biomass among the three treatments (Fig. 2) the sampling design was varied to capture biomass data in the most efficient way; these sampling designs are described below.

Figure 2. Photographs of the secondary peat swamp forests at different stages of forest recovery after fires: (A) long unburnt forests, (B) forests burnt in one recent fire showing vigorous regrowth; (C) forests burnt in two consecutive fires.

4.3 Sampling design

4.3.1 Secondary long unburnt (SF) peat swamp forests

Trees with a diameter at breast height (DBH) ≥ 10 cm, were measured in 10 m radius circular plots, with the more numerous small trees (DBH < 10 cm) measured within a 3-m radius sub-plot, following a protocol developed by Kauffman et al. (2016). All species names and live or dead status were recorded. Dead trees were assessed for the presence of leaves and branches. Coarse woody debris (CWD), defined as detached woody material with diameter ≥ 2.5 cm lying on the forest floor, were measured along a 50-m transect extending through the plot centre following the methodology of Van Wagner (1968). The diameter of CWD intersected by the transect was measured at the point of intersection with the transect, and placed into one of three classes: sound, rotten (signs of decomposition extended to heartwood) or charred (heavily charred wood). Three representative samples for each class were taken for wood density analysis in the laboratory at the facilities of FORDA, Bogor, Indonesia.

Ground cover (i.e. grasses and small shrubs to 0.5 m high), and litter (i.e. leaves, tree fruits, decomposed organic matter and twigs with d < 2.5 cm) were destructively sampled from within a 0.1 m^2 metal frame placed on the peat surface, with a sample taken from north, east, south and west points of the plot circumference (Fig.3A).

A sample of the top peat layer (0-10 cm) was collected from the plot centre point using a 10 cm tall metal cylinder (465 cm^3), with two samples per plot collected for BD and C_{PEAT} analyses.

Samples of peat were collected throughout the peat profile down to the mineral soil surface, from a location near the plot centre point. The following depths were sampled: 10-50 cm (a peat sample for BD and C_{PEAT} analyses was taken from the middle at 20-30 cm depth), 50-100 cm (a sample was taken at 60-70 cm depth), 100+ (a sample was taken midway between 100 cm and the mineral soil surface). Peat was collected using an Eijkelkamp peat sampler (sample length 50 cm; sample diameter 52 mm) attached to the Edelman auger with extension rods enabling sampling up to 6 m (https://en.eijkelkamp.com/products/augering-soil-sampling-equipment/peat-sampler.html). The depth of peat was estimated from the length of the peat auger (rods plus sampling head) inserted to the mineral soil surface (Fig. 3).

4.3.2 Secondary peat swamp forests burnt in one recent fire (SF_1)

A belt transect was selected for sampling the dense vegetation regrowth in this treatment, where moving about on the site was difficult (Fig. 2). Trees were measured along a 50 m transect at sub-plots established at 10 m intervals. Trees were measured within each 1 m radius sub-plot established on both sides of the transect (Fig. 3B), yielding 12 sub-plots per transect. Small trees were counted and placed in the following diameter categories: 0-1 cm, 1-2 cm, 2-3 cm, 3-4 cm, 4-5 cm. Live and dead trees with DBH ≥ 5 cm were measured. Dead trees were assessed for presence of leaves and branches.

CWD was measured at each point of intersection with the 50 m transect and CWD status was recorded. Ground cover was sampled from the sub-plots at the 10 m, 20 m, 30 m and 40 m on the transect (4 samples), followed by collection of litter from the same locations (Fig. 3B). Peat was sampled at the beginning of the transect in the same manner as described above.

4.3.3 Secondary peat swamp forests burnt in two recent fires (SF_2)

A centre-point method (Mitchell, 2015) was chosen to measure trees in this treatment. Scattered dead standing trees across the area, and minor regrowth, meant that neither belt transect, nor circular plot design would adequately capture the distribution of live and dead trees in this treatment (Fig. 2). To capture measurements of all biomass components, three 50 m line transects were established at 60° angle to each other to create a triangle, with each apex representing a centre point (CP) as shown in Fig. 3C. At each CP, the area was visually divided into 4 quarters and one nearest live and one nearest dead tree was measured for DBH and the distance to the CP, resulting in 24 tree records (3CPs x 4 quarters x 2 trees). The distance was measured using a Vertex III (Haglof, Sweden). Live or dead status and species name were recorded and the presence or absence of leaves and branches on dead trees was recorded. On the occasions where no dead trees were present in a 15-20 m distance from the centre point, a count of zero dead trees was recorded.

One of the 50 m transects was chosen for CWD measurements. Ground cover and litter were collected at four locations within the triangle. Peat was collected in the middle of the triangle (Fig. 3).

4.4 Pyrogenic carbon (PyC)

A metal frame of 0.1 m^2 was randomly located on the bare ground of a plot and percent cover of PyC was visually assessed following the Braun-Blanquet (1932) cover-abundance scale (Fig 3). Visually identifiable PyC pieces were collected in plastic zip bags for subsequent dry mass and carbon content analysis in the laboratory; four samples per plot were collected.

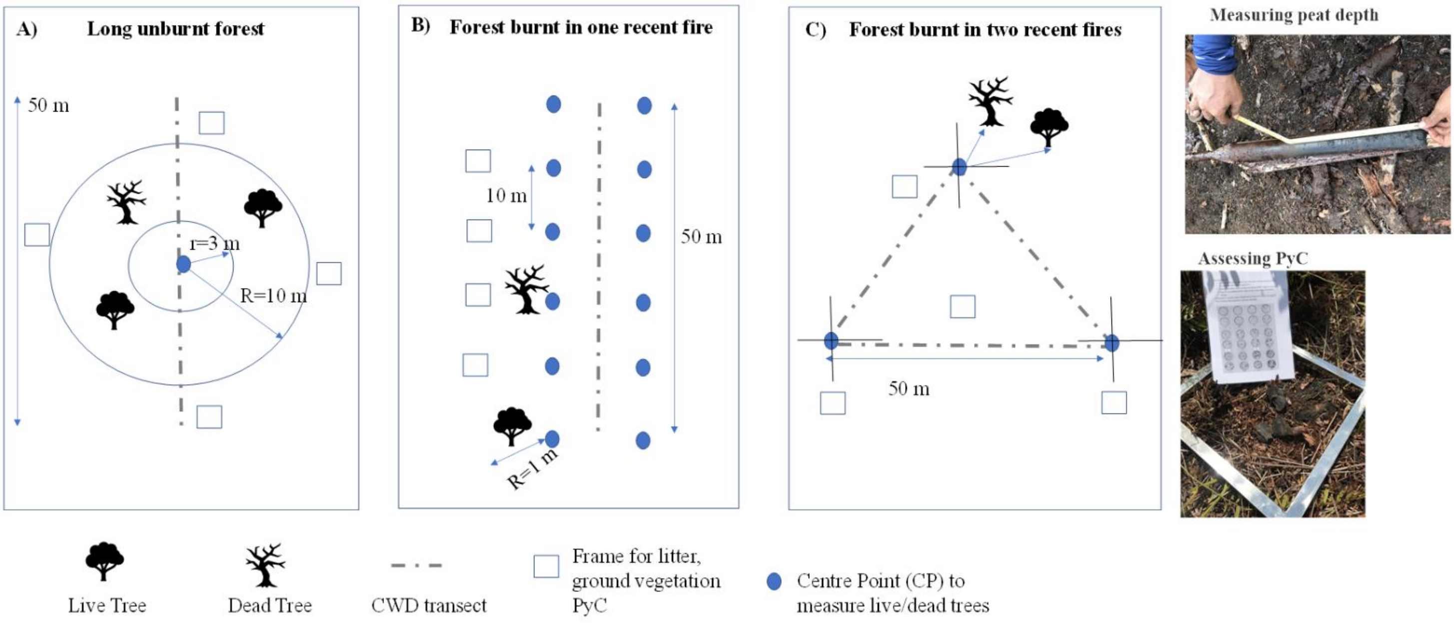

Figure 3. Sampling design

4.5 Estimating aboveground live, deadwood and peat carbon

Live tree biomass was calculated using an allometric equation derived for mixed species of Indonesian peat swamp forests, based on the destructive sampling of 148 trees (Manuri et al., 2014). Dead tree biomass was calculated in the same manner as for live tree biomass but adjusted for the absence of leaves and branches by either reducing the biomass by 2.5% (for a minor defoliation) or 20% (where no leaves, branches or tops were present) (Kauffman et al., 2016). Total tree biomass per hectare (Mg ha^{-1}) for the centre-point method (SF_2 treatment) was estimated as the sum of individual tree biomass (kg) divided by the ¼ circle area (m^2), where the distance from a tree to the CP was the plot radius.

4.6 Sample analysis

Ground cover, litter and PyC were oven-dried at 60ºC for about two weeks to a constant weight in the laboratory of FORDA, Bogor, Indonesia. All measurements are given on a dry-weight basis. For BD estimation peat was oven-dried at 105ºC until dry mass was recorded constant (about 2-5 days) following the protocol of Kauffman et al. (2016). Wood density of CWD was estimated using water a dispersion method. Peat, CWD, and PyC were analysed for carbon content at the facilities of an

ISO/National Certification (KAN) Centre for Agricultural Land Resource Research and Development using a loss on ignition method. The carbon content of live trees was assumed to be 0.47 (IPCC, 2006b).

4.7 Carbon pools

We followed the IPCC (2006a) definition of carbon pools where live aboveground carbon (AGC_{LIVE}) included live trees and ground cover (i.e. grasses and small shrubs); Deadwood included dead standing tees and CWD; Litter; peat carbon and PyC (this carbon pool is not included in the current IPCC methodology, IPCC 2019). Total aboveground carbon (AGC_{TOTAL}) was estimated as the sum of AGC_{LIVE}, deadwood, litter and PyC.

4.8 Power analysis

A range of power analyses were tested to determine the sample size required to reject the null hypothesis (i.e. no difference in the means of carbon pools of long unburnt vs recently and repeatedly burnt forests) with at least 80% power and the probability of a type-I error (α) of 0.05. We defined adequate power using an 80% threshold as this is a common practice in ecology, although we acknowledge that higher power may be desirable in some cases (Di Stefano, 2003). We present power and samples size values beyond 80% so that readers can choose their own threshold.

Due to the spatially hierarchical nature of the study design we applied a linear mixed effects model and conducted the power analyses using simulation following the method developed by Green and MacLeod (2016). A linear mixed-effects model (LMM) was applied to detect the contrast between long unburnt forests and the other treatments, where *block* was a random factor. The effect size was considered individually for each of the major carbon pools (AGC_{LIVE}, deadwood, peat BD 0-10cm) and defined as differences between the SF – SF_1 and SF – SF_2 treatments. To explore trade-offs between sample size and power, we set the analysis to a range of sample sizes extending our simulations by adding more blocks (from 2 to 6) and by increasing the number of plots within blocks (from 3 to 20). Calculations were based on 1,000 Monte Carlo simulations for each combination of sample sizes and contrasts. The calculations were made using software R 3.6.3 (R Core Team, 2020), packages lme4 (Bates et al., 2015) and simr (Green and MacLeod, 2016).

5. Results

5.1 Aboveground carbon

Both fire frequency and time since fire had a major impact on the distribution of carbon across aboveground pools among treatments. Live trees accounted for 91% of the AGC_{TOTAL} in long unburnt forests (SF), virtually disappearing with increased frequency of fire. The contribution of ground cover to AGC_{TOTAL} increased with frequency of fires from 0.7% in long unburnt forests to 7% in repeatedly burnt forests (Table 2). Deadwood was a major carbon pool in forests affected by one and two recent fires (68-75% of the AGC_{TOTAL}), with the majority accounted in CWD (Table 2, Fig. 4). Charred CWD was absent from long unburnt forests and accounted for 6% of total CWD mass on sites burnt in one recent fire, increasing to almost 50% of CWD on repeatedly burnt sites (Fig. 5). The density of CWD (and C) ranged from 583 kg m^{-3} (0.52) for sound, 416 kg m^{-3} (0.46) for rotten and 579 kg m^{-3} (0.66) for the charred class.

For all treatments the litter pool, with an average carbon content around 0.52, accounted for between 2% and 5% of the AGC_{TOTAL} mass – consistently a minor component of AGC_{TOTAL} (Table 2, Fig 4.). PyC was not visible on the forest floor/peat surface in the long unburnt forests. For forests burnt in one recent fire, PyC covered 25±11% (±95% CI) of the forest floor, increasing to around 46% ± 5% at multiple burn forests. The average C% of PyC was 63.3±1.7% (± s.e. of the mean). The contribution of PyC to the AGC_{TOTAL} increased with fire frequency from 3% (or 4.5 Mg C ha^{-1}) after one fire to 12% (or 7.1 Mg C ha^{-1}) after multiple fires (Table 2, Fig. 4).

Overall, the long unburnt forests stored more aboveground carbon than forests affected by recent fires; by comparison one recent fire reduced this by about 20% of the AGC_{TOTAL}, with repeated fire consuming a further 55% of the AGC_{TOTAL} (Table 2).

Table 2 Carbon pools contributing to fires in % of the total AGC in long unburnt forest, SF; forests burnt in one recent fire, SF_1 and forests burnt in two consecutive fires, SF_2. Absolute values in Mg C ha^{-1} are given in brackets.

Carbon pool	Treatment		
	SF	SF_1	SF_2
AGC_{LIVE}	92 (150.4)	26 (34.4)	8 (4.5)
Live trees	91	24	1
Ground cover	1	2	7
Deadwood	6 (10.7)	68 (87.6)	75 (44.1)
Dead trees	3	15	12
CWD	3	53	63
Litter	2 (3.2)	3 (3.4)	5 (2.8)
PyC	0	3 (4.5)	12 (7.1)
AGC_{TOTAL}	100 (164.3)	100 (129.9)	100 (58.5)

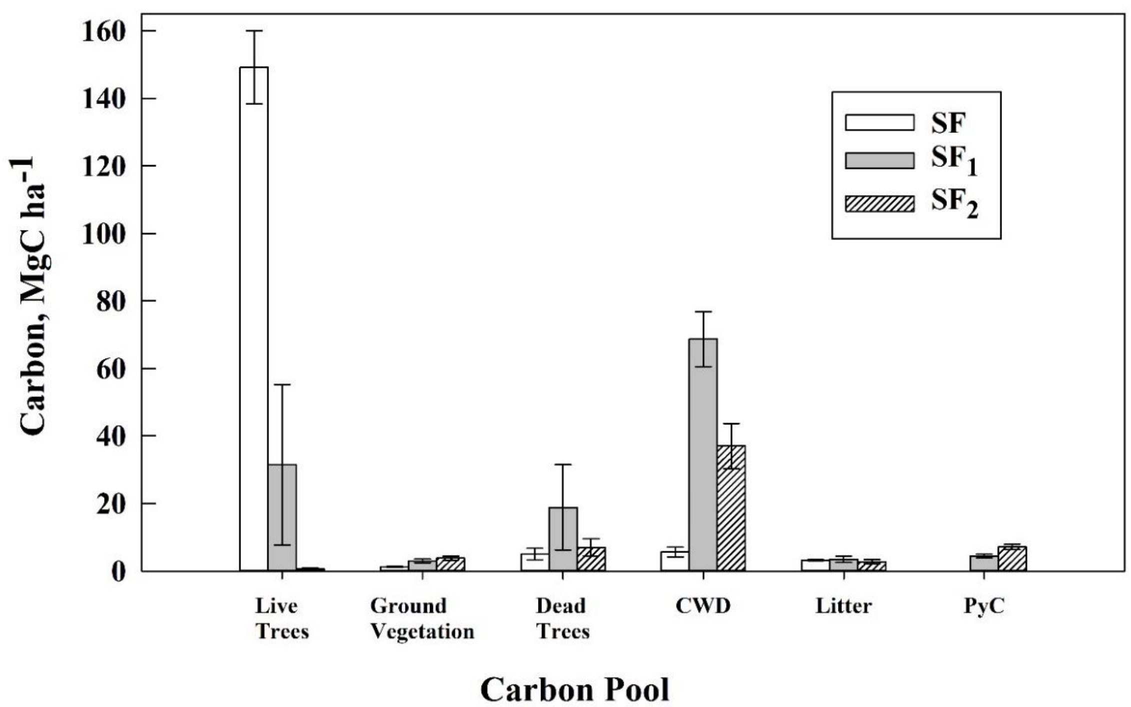

Figure 4. Carbon pools in long unburnt forest, SF; forests burnt in one recent fire, SF_1 and forests burnt in two consecutive fires, SF_2. Values are means, n=6, Error bars indicate standard error of the mean.

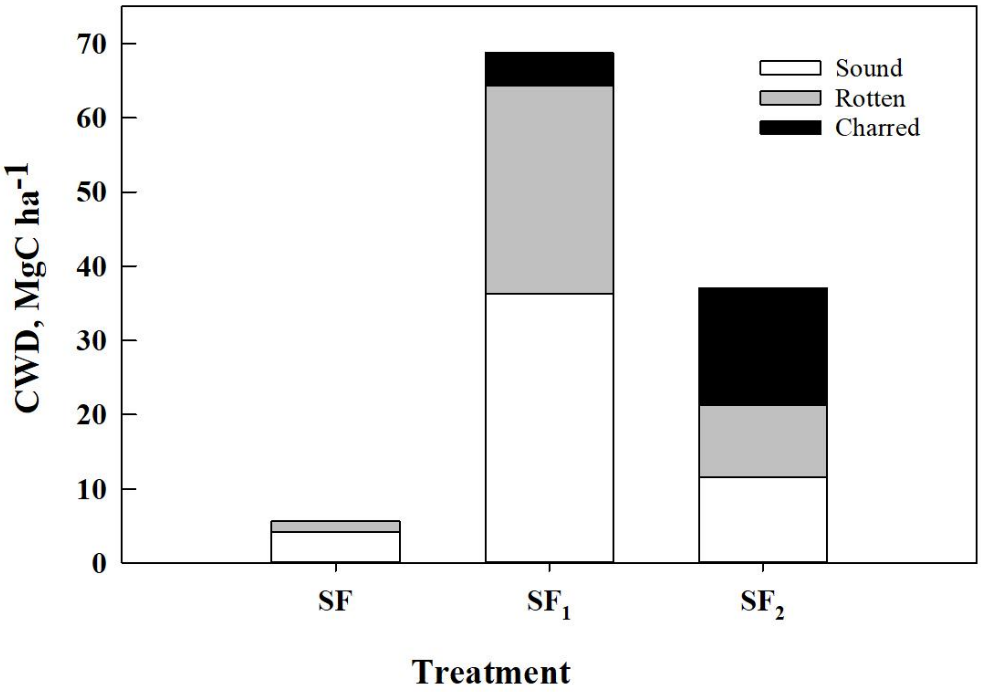

Figure 5. Loads of Coarse Woody Debris (CWD) by the decay class in long unburnt forest, SF; forests burnt in one recent fire, SF_1; and forests burnt in two consecutive fires, SF_2. Values are means, n=6.

5.2 Peat carbon

The depth of peat to a mineral substrate was comparable among the treatments, varying from around 3.5 to 3.65 m (Table 3). Peat BD in the 0-10 cm depth increased with fire frequency (Table 3). The C_{PEAT} increased with increasing frequency of fires from 0.3 in long unburnt forests to 0.37 after one recent fire and to 0.40 after two consecutive fires (Table 3). Reflecting the trend in C_{PEAT}, total peat carbon increased from 2156 Mg C ha^{-1} to 2826 Mg C ha^{-1} with fire frequency (Table 3).

Table 3. Peat characteristics

Treatment	Peat depth, m	Peat bulk density (BD), g cm^{-3}				Peat Carbon content (C_{PEAT}), %					Total, Mg C ha^{-1}
		0-10 cm	*10-50 cm*	*50-100 cm*	*100+ cm*	*0-10 cm*	*10-50 cm*	*50-100 cm*	*100+ cm*	***average***	
SF	3.49±0.12	0.124±0.01	0.221±0.29	0.293±0.08	0.216±0.02	42.9±3.5	28.2±5.5	17.6±2.5	29.2±7.2	29.5±3.5	2156±208
SF_1	3.66±0.09	0.136±0.01	0.221±0.02	0.205±0.02	0.205±0.01	41.7±1.5	28.5±7.4	40.0±0.2	38.9±2.2	37.3±2.3	2511±100
SF_2	3.64±0.05	0.154±0.02	0.198±0.02	0.172±0.01	0.164±0.01	41.4±1.3	39.5±2.6	37.9±1.5	41.5±0.5	40.3±2.3	2826±162

Values are the mean ± s.e. of the mean. N=6 for peat bulk density; N=3 for C%. SF is Long unburnt forests; SF_1 – Forest burnt in one recent fire, and SF_2 – Forests burnt in two recent fires

5.3 Power analysis

For determining the optimum sample size, we considered the effect size based on the data variability and the contribution of the individual carbon pools to the emissions estimates. We observed a significant effect of fire frequency and fire return interval of AGC_{LIVE} carbon and assumed that a 30% (or 45 Mg C ha^{-1}) difference in the means would be a reasonable effect size. A 30% difference in AGC_{LIVE} at 80% power can be detected by either increasing the number of plots within a block or by increasing the number of blocks and reducing the number of plots within blocks – each solution results in 12 plots per treatment arranged as either 2 blocks x 6 plots, 3 blocks x 4 plots or 6 blocks x 2 plots (Fig. 6).

Peat BD is an important parameter contributing to the estimates of mass of peat, thus it is important to make an effort to accurately characterise it. For power analysis we focused on the top layer (0-10 cm), for which we had the most accurate sampling. Based on the predicted means, we considered that a difference of 0.06 g cm^{-3} (or 50%) would be a reasonable effect size. The simulations revealed that sample size reduces with the increased number of groups (blocks), similar to the trend observed for AGC_{LIVE} (Fig. 6).

Deadwood was the most variable carbon pool, mainly due to a high variability in dead standing trees in recently burnt SF_1 forests (Fig. 4). Because the long unburnt forests stored only a minor fraction of carbon in deadwood (10.8 Mg C ha^{-1}), setting up an effect size of 30-50% difference would require detecting a difference of 3-5 Mg C ha^{-1}, which is unrealistically small considering that recently burnt forests stored 40-80 Mg C ha^{-1} in deadwood (Figs. 4-5). Thus, for the deadwood, the effect size was set up at the difference of 100% or 22 Mg C ha^{-1}, and even with such a rather large effect size, our power curves indicate that the sample size must be in order of 26-30 plots per treatment to yield greater than 80 % power (2 blocks x 13 plots, 3 blocks x 9 plots or 6 blocks x 5 plots; Fig. 6). Increasing the effect size to 150% (or 33 Mg C ha^{-1} difference) produced the same results as for the other two pools (12 plots per treatment, data not shown).

No power analysis was conducted for PyC as this carbon pool is not present on the forest floor of long unburnt forests, thus any combination of block and plot sample sizes discussed above will be able to detect a significant difference between the treatments.

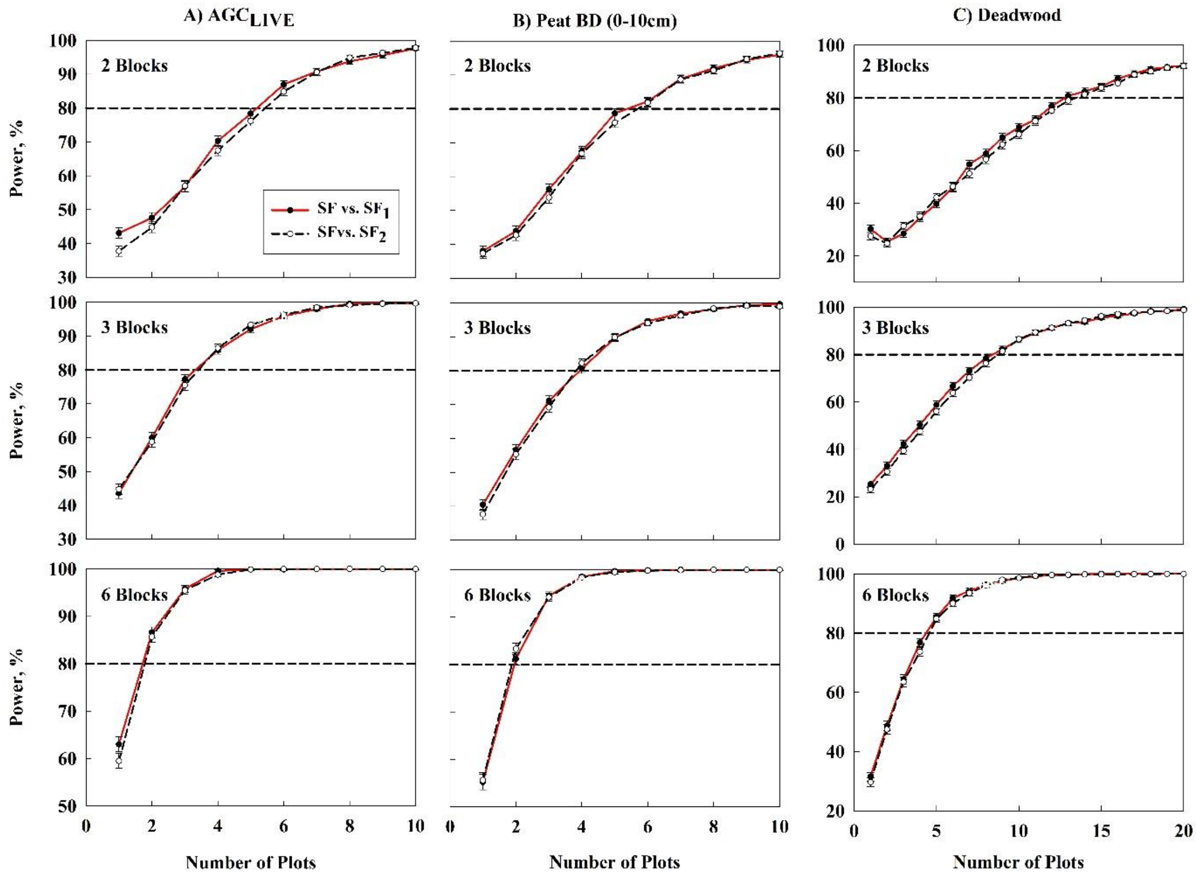

Fig. 6. Power analysis of the major carbon pools using a combination of blocks and plots to achieve 80% power to detect A) a 30% difference in the means for aboveground pools (live trees and ground vegetation); B) a 50% difference in the means for peat bulk density 0-10cm depth and C) a 100% difference in the means for deadwood (dead trees and CWD) between recently burnt forests vs long unburnt forests. Values are the means based on 1,000 simulations; error bars are the standard error of the mean.

6. Discussion

6.1 Impact of fire frequency on aboveground carbon

In this study we present a comprehensive assessment of the aboveground and peat carbon pools as they are affected by recurring fires. The review of literature showed there is great uncertainty in the estimates of peat fire emissions, especially where peat swamp forest sites are burnt in more than one fire. This pilot study shows that after one recent fire about 90 Mg C ha^{-1} remains aboveground as the deadwood carbon pool (Figs. 1, 4, Table 2), similar to findings of others (Qirom et al., 2018; Siahaan et al., 2020). Following a second consecutive fire, about a half of the deadwood is retained, mainly as CWD, or converted to pyrogenic carbon. Overall, only about one third of the total aboveground carbon in the form of deadwood and PyC remains after two consecutive fires, compared to the long unburnt forests, which mainly stores aboveground carbon in live trees. Both deadwood and PyC are burnt mainly in smouldering combustion that releases an array of potent greenhouse gases (Andreae and Merlet, 2001; Stockwell et al., 2016) so that excluding these fuels from carbon mass balance would lead to greater uncertainties in the emission estimates.

We did not observe the impact of fires on litter, with predicted means being similar for all of the treatments ranging from 2.77 (SF_2) to 3.4 (SF_1) and 3.2 (SF), findings similar to other studies for peat swamp forests of Indonesia (Qirom et al., 2018; Siahaan et al., 2020). Considering that litter contributes a rather small fraction of the carbon emitted from peat fires, we advocate that for accounting of this carbon pool in emissions estimates, an average number can be considered.

6.2 Role of the pyrogenic carbon in the emission estimates

The role of PyC in forest carbon balance and its contribution to emissions is largely ignored due to a lack of empirical data; as such the PyC data presented here is novel in addressing this important knowledge gap. Consistent with studies from cold temperate peatlands (Worrall et al., 2013), we observed that one fire produced PyC equivalent to about 3% of aboveground biomass and that repeated burning increased this contribution threefold. Clearly PyC becomes an increasingly important carbon pool in repeatedly burnt peat swamp forests. Ignoring fire produced PyC from carbon mass balance will lead to overestimation of atmospheric emissions, with recent studies pointing to the overestimates of 4% of the global total emissions (Surawski et al., 2016). This study was not designed to estimate emissions from peat fires but rather to address the lack of a more complete knowledge of parameters required for improved peat fire emission estimates. As such we refrain here from giving examples of the difference in emissions estimates where all aboveground fuels and PyC are considered vs peat only released emissions – a subject for follow up studies.

6.3 Peat carbon

In these peat swamp forests, peat to the soil mineral surface stored in excess of 2000 Mg C ha^{-1}, or about 2-7 times more than was stored aboveground. The difference in peat carbon stocks between treatments mostly reflected variability in peat BD and C_{PEAT} among the treatments. While we are cautious that peat data from our pilot study is based on a limited number of samples, the trend of increased carbon concentration with increased frequency of fire is similar to findings from temperate and boreal needleleaf forests, but in contrast to observations from frequently burnt savannas and broadleaf forests (Pellegrini et al., 2018). Often the effect of fires on soil carbon is overlooked because samples are collected shortly after fires (Santín and Doerr, 2016). In this study the peat samples were collected several years after the fires and it is possible that increased C_{PEAT} reflects eluviation of fine PyC from the peat surface down into the peat profile. Similar to findings of Sinclair et al. (2020), we observed an increase in *BD* in the top layer (0-10 cm) with increasing frequency of fires. Peat degradation typically increases the bulk density of peat soil through drainage, heating of the peat surface or from compaction (Ali et al., 2006; Hooijer et al., 2012). Because the C_{PEAT} and *BD* parameters determine M_{PEAT} (Eq. 1), they should be the focus of further studies to improve emissions estimates from repeatedly burnt forests. Moreover, the *BD* should vary with fire frequency in a similar manner as peat depth, a suggestion echoing Konecny et al. (2016).

The overall difference in peat depth (*h*) between long unburnt, recently and repeatedly burnt forests was minor and we understand that it was mainly related to the position of plots on the *Mawas* peat dome (Page et al., 1999); that is, to the elevation of the peat surface and the peat base (Silvestri et al., 2019) as well as sampling peat depth with an auger (when it is difficult to pick up a small difference in peat consumption several years after a fire). A more careful sampling of peat loss during fires is required to improve knowledge for the *h* parameter.

6.4 Sampling design and power analysis

We developed field sampling designs to quantify biomass pools under the contrasting field conditions of our treatments (long unburnt, recently burnt and repeatedly burnt peat forests, Fig 2). A plotless design was applied to the repeatedly burnt SF_2 treatment to capture the different distribution of live and dead trees in this treatment relative to the other treatments. The average distance from the centre point to live trees was 8 m (range 0.54 - 38 m) and to dead trees was 15 m (range 1.75 - 29.6 m), meaning that more traditional plot shapes (circle, square, rectangular) would need to be either very large (thus very time consuming in measuring all trees), or would not be able to capture the observed variability if plot radius is kept to 10 m as for SF treatment. While the sample plots varied in shape from belt transect (SF_1), to circular plot (SF) to quarter-center (SF_2), each comprised a systematic random sampling with the same threshold for measuring carbon pools, making it an unbiased design. For the analysis of data derived from such designs, it should be considered that response variables may not share common residual variances. Therefore models accounting for non-homogeneous errors should be implemented (Harrison et al., 2018).

In this study we applied a power analysis to ensure that future experiments can be designed in a way to adequately test for the effect of fires on peat forest carbon pools with at least 80% probability. Power analysis is a useful tool for investigating the effectiveness of different sampling designs and depending on the structure of the variance, power to detect trends may be increased by altering the sampling design (Perles et al., 2014). We considered a few combinations of *along* and *within* argument by extending the number of groups (blocks) and the number of observations (plots). Depending on the specific objectives, resources and ability to spatially fit the required number of plots within a block, or the required number of blocks within the treatment, various combinations of 'fewer blocks - more plots' or 'more blocks - fewer plots' can be selected with similar power. Considering the challenging field conditions of Indonesian peatlands and the time and effort required to move between blocks and plots, we would recommend a middle approach: fewer blocks (3-4) and more plots (5-6), as the distance between plots is generally shorter than the distance between blocks – yet a greater than two number of blocks would better capture variability of forest carbon among pools and treatments.

For the effect sizes considered in this study, except for the deadwood, the power to detect change was very high using just a total 12 plots per treatment (in any variation of blocks and plots). For deadwood, the sample size was much greater because the magnitude of detectable changes was smaller, confirming other studies that a small change is difficult to detect with high power (Westfall et al., 2013). Increasing the effect size in the deadwood pool led to similar results as for other pools. It is important to remember that our power analysis is only an approximation of a desired sample size as the outcomes were influenced by the variability in our data, desired effect sizes and the α threshold; changing any of these parameters would alter the outcomes as was observed for deadwood. We based our power analysis on a reasonable detectable change which would make an impact on emissions estimates. To this end we conclude that a minimum of 12 plots per treatment is a good starting point for future sampling strategies and field sampling designs.

7. Conclusion

This study identifies current knowledge gaps in supporting data required to improve our understanding of the emissions from Indonesian peat fires. Despite many publications on peat fire emissions, an important knowledge gap in empirical observations to support estimation parameters remains. Lack of knowledge of the impact of repeated fires on aboveground fuels and on the production of pyrogenic carbon adds to uncertainty in emissions estimates. Using the data from our pilot study we show that PyC plays an important role in the carbon balance of peat swamp forest and that its importance increases with frequency of fires. It can be argued that the major source of fire emissions in tropical peat forests is the peat, while the contribution of aboveground fuels is relatively minor (Tables 2-3), yet to improve the accuracy in peat fire emission estimates and carbon mass balance, we would argue that all aboveground and peat carbon pools must be properly accounted for. This study aimed to develop an appropriate sampling design and to estimate the required sampling intensity for describing and comparing emissions from peat swamp forests at different stages of degradation. As such we refrained from a comprehensive statistical analysis and from providing recommendations for refining peat fire emissions estimates. This study provides crucial information for the design and implementation of further field experiments to evaluate the effect of repeated fires on aboveground carbon pools and peat for reducing uncertainty in peat fire emission estimates.

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Declaration of interest

Authors declare no competing interests

Authors contribution

LV, HK, CJW conceived and designed the study, overseen its implementation and contributed to the development of the manuscript idea. LV has analysed the data, written the first draft of the manuscript and conducted literature review with the inputs from HK and CJW. HK overseen logistic of the field campaign. WCA led field sampling and sample analysis, collided the data and conducted data quality control. RI led study sites selections and fire history analysis. All authors participated in field data collection, contributed to data analysis, and writing of the manuscript.

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