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Variable carbon losses from recurrent fires in drained tropical peatlands

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Carbon losses from tropical peatland fires

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Abstract

Tropical peatland fires play a significant role in the context of global warming through emissions of substantial amounts of greenhouse gases. However, the state of knowledge on carbon loss from these fires is still poorly developed with few studies reporting the associated mass of peat consumed. Furthermore, spatial and temporal variations in burn depth have not been previously quantified. This study presents the first spatially explicit investigation of fire-driven tropical peat loss and its variability. An extensive airborne LiDAR (Light Detection and Ranging) dataset was used to develop a pre-fire peat surface modeling methodology, enabling the spatially differentiated quantification of burned area depth over the entire burned area. We observe a strong interdependence between burned area depth, fire frequency and distance to drainage canals. For the first time, we show that relative burned area depth decreases over the first four fire events and is constant thereafter. Based on our results, we revise existing peat and carbon loss estimates for recurrent fires in drained tropical peatlands. We suggest values for the dry mass of peat fuel consumed that are 206 t ha^{-1} for initial fires, reducing to 115 t ha^{-1} for second, 69 t ha^{-1} for third and 23 t ha^{-1} for successive fires, which are 58% to 7% of the current IPCC Tier 1 default value for all fires. In our study area, this results in carbon losses of 114, 64, 38 and 13 t C ha^{-1} for first to fourth fires, respectively. Furthermore, we show that with increasing proximity to drainage canals both burned area depth and the probability of recurrent fires increase and present equations explaining burned area depth as a function of distance to drainage canal. This improved knowledge enables a more accurate approach to emissions accounting and will support IPCC Tier 2 reporting of fire emissions.

Introduction

Tropical peatlands store huge amounts of carbon as incompletely decomposed plant material that has accumulated over thousands of years in waterlogged, anaerobic environments. Current estimates indicate that they cover an area in the range of 39–66 million hectares (ha), which is 10–16% of the global peatland resource (Page *et al.*, 2011). As a result, tropical peatlands are one of the largest near-surface pools of terrestrial organic carbon, with a total peat carbon pool of 82–92 Gt, of which 65% is located in Indonesia (Page *et al.*, 2011). In addition to carbon storage, these wetland ecosystems also play a significant role in supporting biodiversity, with a unique combination of habitats and endemic and endangered species (Posa *et al.*, 2011; Wibisono *et al.*, 2011). Further, they provide a wide range of valuable environmental goods and services including forestry and fishery products, flood mitigation and climate regulation (Rieley and Page, 1998).

Over the last two decades, the connected processes of land use change (logging, deforestation, conversion to plantation estates and small-holder agricultural plots), drainage and fire have contributed to loss and degradation of this ecosystem, resulting in increased greenhouse gas (GHG) emissions (Page *et al.*, 2002; Ballhorn *et al.*, 2009; Hooijer *et al.*, 2010, 2012; Jauhiainen *et al.*, 2012), local and regional air pollution (See *et al.*, 2007; Heil & Goldammer, 2001), a severely reduced biodiversity (Posa *et al.*, 2011) and loss of local livelihood opportunities (Chokkalingam *et al.*, 2005). In their natural state, these peatlands have a high water table, at or just below the peat surface for most of the time, and are covered by forest (Rieley & Page, 2005). Disturbance or loss of the forest canopy and persistent lowering of the water level promote an enhanced rate of aerobic peat decomposition (Hooijer *et al.*, 2012; Jauhiainen *et al.*, 2012) while also greatly enhancing the risk of both vegetation and peat fires (Langner *et al.*, 2007). In Southeast Asia, nearly all peatland fires are of anthropogenic origin, started by farmers or private companies as well as government agencies on both small and large scales as part of livelihood activities or to facilitate the conversion of forest land to plantations or timber estates (ADB/BAPPENAS, 1999; Bompard & Guizol, 1999; Bowen *et al.*,

2000; Siegert *et al.*, 2001). Under certain conditions (e.g. where peatlands have been subject to disturbance and drainage, and during dry weather conditions), fires set to clear vegetation can ignite the peat, resulting in long-lasting and smoldering fires that release large amounts of carbon to the atmosphere (Page *et al.*, 2002; Rein *et al.*, 2008; Ballhorn *et al.*, 2009; Langner & Siegert, 2009; Watts & Kobziar, 2013). Since the 1997/98 El Niño-induced drought and associated widespread fires, tropical peatland fires, especially in insular Southeast Asia, have recurred on an almost annual basis during most dry seasons, causing on-going loss of stored carbon, as well as heightened tensions between neighbouring countries (van der Werf *et al.*, 2008; Gaveau *et al.*, 2014).

The current scale of carbon emissions from peatlands in Southeast Asia, whether from biological oxidation or fire, has increased interest in tropical peatlands in the context of global warming (Page et al., 2002; Hooijer et al., 2010), but any attempt to secure financial support for emissions reduction (e.g. through REDD+ or other carbon market schemes) requires reliable methodologies that can measure, report and verify (MRV) GHG emissions on a regular basis before, during and after any emissions mitigation intervention has been undertaken. The IPCC publication "2013 Supplement to the 2006 IPCC Guidelines for National GHG Inventory: Wetlands" (Drösler *et al.*, 2014) provides the basis for the development of a GHG emissions accounting methodology for fires on drained organic soils. Required parameters for quantification of fire-driven carbon loss are area of peatland burnt, mass of dry matter (peat fuel) available for combustion, and combustion factor. The mass of fuel available is derived from measurements of depth of burn and peat bulk density which, in combination with data on peat carbon content, can be used to calculate the loss of carbon during combustion, or with emissions factors, to calculate the gaseous products of combustion (Drösler *et al.*, 2014). For detailed emission estimates (e.g. at Tier 2), data on the variation in the mass of fuel as well as regional factors for stratification (e.g. by fire frequency or drainage intensity) need to be incorporated. Knowledge gaps in these areas (in particular, spatial and temporal variations in depth of burn, but also peat bulk density and carbon content) contribute to the overall uncertainties

related to the scale of emissions from peatland fires. Due to the often restricted accessibility, especially in remote locations, and the difficulties in approaching burning peat fires in the field, there are only a very limited number of studies providing ground-based data on the mass of peat consumed during tropical peatland fires (Page *et al.*, 2002; Ballhorn *et al.*, 2009). Remote sensing approaches are an efficient tool to overcome these problems. Airborne Light Detection and Ranging (LiDAR) technologies, specifically, can provide terrain surface elevation information with a geometric accuracy in the range of decimeters to centimeters depending on vegetation type and terrain conditions. LiDAR is based on the transmission of laser pulses towards the ground and the detection of the backscattered radiation. By measuring the time delay between the transmission of a pulse and the recording of the return signal, surface elevation can be derived. Reddy et al. (2015) used pre- and post-fire LiDAR data to estimate elevation change from a temperate peatland fire with an average elevation loss of 0.46 ± 0.18 m. Ballhorn et al. (2009) used LiDAR to determine depth of burn for single fire events in peat swamp forests in Central Kalimantan, Indonesia, by measuring the elevation difference at the border between burned and unburned peatland. On average a mean burned area depth of 0.33 ± 0.18 m was observed. Both studies neither investigated the impact of repeat fires nor proximity to drainage features. Assuming that both variables have an interdependent effect on the lowering of the peat surface in burned areas, the uncertainty of carbon loss estimates will be reduced by analysing them in a space time domain.

The main objectives of the current study were i) to investigate and hence to better understand the dependence of total and relative burned area depth on fire frequency and distance to drainage canals, ii) to define burned area depth as a function of these factors and iii) to produce revised carbon loss estimates for recurrent fires on tropical peatlands.

In our study we define burned area depth as the difference between pre- and post-fire terrain elevation, where we use the term burned area depth as the total cumulative depth measured. Relative burned area depth is referred to as the lowering of the peat surface after a single fire event

in comparison to the terrain elevation before that fire, i.e. including a preceding lowering in the case of two or more fires. Relative burned area depth values are derived from the total burned area depth measurements.

Materials and methods

Study area

Our 220,000 ha study site (Fig. 1) is located in a peatland dominated landscape in the Kapuas District of Central Kalimantan, Indonesia, which is part of the former Mega Rice Project (MRP). The MRP was a resettlement project initiated by the Indonesian government in 1995, which was ended in 1998 when it was recognized to be failing (Muhamed & Rieley, 2002). The associated construction of more than 4,000 km of drainage canals led to serious degradation of more than one million hectares of peatland. Additionally the area was and is subject to both legal and illegal logging activities since the mid 1980s.

Using historical Landsat imagery and recent RapidEye scenes we determined that nearly 50% of the 220,000 ha had been affected by up to eight fire events between 1990 and 2011 (Fig. 1 and 2).

An analysis of NOAA AVHRR hotspots (Langner & Siegert, 2009; Li *et al.*, 2003) and Terra/Aqua MODIS active fire data (Giglio *et al.*, 2003) in relation to their distance to drainage canals (Figs. 3a and 3b) confirmed previous results indicating that the likelihood of fire was increased by clearance and drainage (Page *et al.*, 2002), even 15 years after drainage had commenced.

LiDAR point cloud filtering

The 700,000 ha small-footprint airborne LiDAR dataset was acquired between 15.08.2011 and 15.10.2011 with an Optech Orion M200 airborne laser scanner at a nominal altitude of 800 m above ground with an average point density of 2.8 points per m². Ground and off-ground points were classified by applying a hierarchic robust filtering to the LiDAR point cloud. This method is based on

point cloud pyramids from coarse to fine resolution and iteratively interpolates the Digital Terrain Model (DTM) by using linear prediction (Kraus, 1998). In the first iteration, a rough approximation of the surface is computed starting at the coarsest pyramid level with all points being equally weighted. Under consideration of the weight function calculated from the residuals of the points, the surface is iteratively recomputed until a stable situation is reached. The DTM is then compared to the data of the next pyramid level and points with a defined tolerance to the surface are used as input for the next iteration (Pfeifer et al., 2001). This process also includes the classification of the point cloud. In order to permit a faster processing we applied this method on a thinned out point cloud where we computed a preliminary DTM with 1 m resolution. Subsequently we reintroduced all points from the original point cloud lying below the DTM and selected the deepest point for each 1 m cell. Based on this dataset we performed a detailed manual quality control where we eliminated all remaining non-ground points (mostly bushes and dead lying trees). The quality control was individually adjusted to the occurring land cover type, e.g. areas with dense low vegetation like ferns and shrubs were filtered more rigorously. Figure 4 shows an example of the filtering result with different vegetation types and fire frequencies. The final DTM was generated from the deepest points within 5 m cells by using a Kriging algorithm (Papritz & Stein, 2002). The vertical accuracy of the DTM was assessed based on 441 checkpoints, which were measured by differential GPS and total station measurements during a field survey conducted in 2010. For the land cover types peat swamp forest and burn scar the DTM had a vertical accuracy of 0.12 m RMSE and 0.19 m RMSE, respectively. For all land cover types identified the modelled surface was within a 95% confidence interval of the true surface.

Burned area delineation

Earth observation based active fire data from the NOAA AVHRR and Terra/Aqua MODIS sensors were analysed for the determination of years with fire occurrence. The AVHRR active fire product (Langner & Siegert, 2009; Li et al., 2003) was used for the years 1997–2000, while the MODIS active fire product (Giglio et al., 2003) was employed for the years 2000–2011. The preliminary hotspot

analysis indicated a selection of the years 1997, 2001, 2002, 2004, 2005, 2006, 2009 and 2011 for burned area mapping. Additionally 1990 was chosen to get a picture of the area before the establishment of the MRP. For the further delineation of the burned areas these fire hotspot datasets were not implemented.

A total number of 49 Landsat-5 TM and Landsat-7 ETM+ scenes (Level-1T) with 30 m spatial resolution from 1990 to 2011 were automatically classified based on object-based classification algorithms. The images were atmospherically corrected with the ATCOR-2 software, which employs the MODTRAN atmospheric transfer model to convert the Digital Numbers (DNs) into surface reflectance (Richter *et al.*, 2006). Recently burned areas (2009–2011) were identified by visual interpretation of high resolution RapidEye images (5 m spatial resolution), acquired on 10.02.2010 and 29.07.2012, as well as the LiDAR DTM. The RapidEye images were geometrically corrected by a semi-automatic image-to-image matching procedure in order to fit the Landsat imagery and atmospherically corrected with the ATCOR-2 software. The edges of automatically classified historic burned areas were refined based on the RapidEye images in areas where they were still clearly visible. The 2011 burned area analysis only considered data acquired before the LiDAR flight campaign. Fire frequency was determined by overlaying resp. intersecting the burned areas for every year of fire occurrence.

Drainage canals and logging tracks identification

Drainage canals and recent logging tracks (skid rails and ditches) were identified by visual interpretation of the LiDAR DTM.

Spatial modelling of the pre-fire peat surface

Comparison of different interpolation techniques

Southeast Asian peatlands are typically located at low altitudes where they have developed smooth convex-shaped domes that feature slight topographic gradients with a rise of only about 1 m per km (Anderson, 1983; Diemont & Supardi, 1987; Neuzil, 1997; Page et al. 1999; Rieley & Page, 2005). This characteristic facilitates spatial modeling of a pre-fire peat surface. Pre-fire elevation in selected burned areas was modelled by spatial interpolation using reference elevation values from surrounding unburned areas. A parametric interpolation technique originating from the field of Computer Aided Geometric Design (Bézier approximation), as well as five methods commonly used in geosciences (Inverse Distance Weighted, Natural Neighbour, Trend, Spline and Kriging) were tested against an unburned model site of 10,000 ha with a simulated burned area (approx. 2,000 ha) at its center. This approach allowed the evaluation of the modeled surface not only in those areas used as reference for interpolation, but also within the extent of the excluded area, which is usually not known in the operational application of the method. Values in Table 1 show the error statistics related to the difference between modeled surface and DTM. They are based on a validation set of 100,000 random points divided into two classes depending on their location in the test site (reference area or interpolated area). Mean and standard deviation of the errors in the interpolated area are lowest in the case of Bézier approximation. Minimum and maximum of the interpolated area are in the same dimension as in the reference area and arise from the natural heterogeneity of the peat surface rather than from outliers in the model.

We additionally tested our approach based on the filtered LiDAR ground points in order to exclude any potential bias referable to the DTM interpolation. Since no significant difference was observed and, operationally, it is more efficient to use values derived from the DTM, we decided to further use the approach based on the DTM.

Implementation of Bézier approximation

According to the interpolation results of the different methods (Table 1), Bézier approximation was implemented for the pre-fire peat surface reconstruction (5 m spatial resolution). Bézier surfaces are a straightforward extension to Bézier curves. With the use of the Bernstein binomial coefficients a Cartesian product is applied to the equations of the parametric curves (Salomon, 2006). In general the number of reference points defines the polynomial order of a Bézier surface. In order to model a continuous surface based on thousands to millions of points an alternative approach is necessary. For this purpose a least squares method for estimating Bézier surfaces, presented by Engels (Engels, 1986), was adapted.

We applied Bézier approximation to two test sites of approx. 15,000 ha (Block A/E) and 22,000 ha (Block B), in which approx. 1,700 and 2,300 ha had been affected by up to seven fire events (Fig. 1). Requirements for their selection were twofold: Firstly, the whole extent of the site had to be situated completely on peat since other landscapes have different surface characteristics that, in the case of an irregularly undulating terrain, cannot be appropriately described with the limited number of parameters of the Bézier method. Secondly, the burned areas had to have an unburned reference area on, ideally, each side which was used for interpolation. Reference points for the interpolation were generated exclusively in areas with a distance of at least 200 m from large drainage canals. Buffer zones of 20 m around logging tracks and 30 m around burned areas were excluded to avoid any impact of potential misclassifications, e.g. introduced by differing spatial resolutions.

Bézier approximation was performed with different polynomial orders. The RMSE was calculated for each model, where the most accurate result was achieved for five consecutive polynomial orders with non-significantly differing values (Table 2). Since reference data of the unburned pre-fire peat surface were not available, outliers of the models within the interpolated area could not be identified based on a single model. In order to compare the five qualified models and to make them

robust against unquantifiable outliers, mean and standard deviation of the five models were calculated. The mean was used for further analyses, where areas with standard deviation higher than the modelling accuracy, i.e. 13 cm in Block A/E and 10 cm in Block B, were excluded, with significant differences occurring solely within burned areas near the edges of the processed areas. The difference between the modelled surface and the LiDAR DTM indicated a measure of burned area depth at each pixel.

For validation purposes, values derived from the peat model were compared to measurements derived directly from the LiDAR DTM. A contour based approach similar to that described in Ballhorn et al. (Ballhorn *et al.*, 2009) was applied to 30 locations representing burned areas that burned only in 2009. Areas 100 m along and 50 m across the burned area borders were analysed by calculating the differences in mean elevation between the burned and unburned sides (10 m zone around the boundary excluded). On average a burned area depth of 0.179 ±0.089 m was determined. For comparison, the difference between normalized peat model values in both directions of the burned area border was calculated with an average of -0.188 ±0.096 m and a non-significant disparity of 0.009 m.

Quantification of burned area depth considering fire frequency and distance to drainage canals

For the quantification of burned area depth a sample based method was applied. Approximately 50 random points per ha with at least 5 m spacing were generated within the burned area. Allowing for the lower spatial resolutions of the satellite images used for burned area delineation, the total extent of the burned area was reduced by a buffer zone of 30 m. Points with a distance to drainage canals of smaller than 200 m were excluded from analyses. Furthermore, only points with a distance equal to or greater than 20 m from small logging tracks were considered in order to assure no points lay directly within these features, potentially resulting from digitisation inaccuracies. Analyses

discriminating between fire frequencies excluded points within a distance of 30 m from fire frequency class borders.

To analyse the relationship between burned area depth and distance to drainage canals the random points within the burned area were classified into 100 m intervals regarding their distance to the nearest canal. Mean elevation differences between the reconstructed peat surface and the DTM were calculated for each distance class. Standard deviation and 95% confidence interval of the mean were determined for validation purposes. Values within each class met the requirement of a normal distribution (Table S1).

For the analysis of burned area depth as a function of fire frequency, the mean differences between modeled surface and DTM as well as associated standard deviations and confidence intervals of the means were calculated for each fire frequency class (Table S2).

In order to investigate the dependence of burned area depth on both the number of fire events and the distance to drainage canals, a multidimensional analysis considering either factor was performed. Logarithmic trend lines were fitted to the total number of sample points of each fire frequency class as functions of the distance to drainage canals with residual standard errors between 0.13 and 0.15 m. Since, collectively, the highest fire frequencies (four and more) occur only up to a distance of 400 m from canals, larger distances were not considered for this class.

Sampling and analysis of peat bulk density and carbon content
Peat characteristics were determined following the protocol described in Hooijer *et al.* (2012).
Peat samples were obtained during the dry season (June–October) of 2012 and 2013 from 18 soil pits located in the study area. Four pits were located in forest with high drainage impact (approx. 50 m from nearest canal) but no fire disturbance and five pits in forest with low drainage impact

(>1000 m from nearest canal) and no fire disturbance. Nine pits were located in burnt peatland with moderate to high drainage impact (100–400 m from canals).

Sampling of peat for bulk density determination was at 0.1 m intervals between 0.1 and 0.4 m. Three replicate samples were taken at each depth, using sharpened steel rings of 8 cm diameter and 8 cm length that were custom made to reduce peat compression. Peat bulk density was determined by drying samples at 105°C for up to 96 hours, with measurements at interim intervals to ensure there was no further weight loss, i.e. no moisture was left in the samples.

Peat samples for carbon analysis were taken from three of the bulk density sampling pits, at 0.2 m below the surface. Two of the pits were in unburnt forest, and one in a burnt area. Nine replicate samples were analysed for each location. Carbon concentration was analysed by measuring weight loss after burning the peat for 6 hours in a muffle furnace at 550°C.

Results

Burned area depth

Interdependency between burned area depth, fire frequency and distance to drainage canals
The results of this study show a strong interdependency between burned area depth, number of fire events and distance to drainage canals. The relationship between burned area depth and the distance to canals (without considering fire frequency) expresses itself in a non-linear decrease of burned area depth with increasing distance from these drainage features (Fig. 5a). An impact of canals on the burned area depth can be observed up to a distance of at least 800 m.

In relation to fire frequency (without considering distance to drainage canals), relative burned area depth decreases for each successive fire event over the first three fires (-0.17 m, -0.10 m, -0.06 m). For locations experiencing four or up to seven fires, the mean relative depth is -0.13 m (Fig. 5b).

Parametric t-tests for unequal variances proved the decrease to be statistically significant with all levels of significance being smaller than 0.001. An investigation of the spatial distribution of each fire frequency class showed that areas with four fires or more occurred only up to a specific maximum distance of 600 m from drainage canals, while locations with less fire events occurred across a wider zone up to 1,300 m from canals. About 60% of the points experiencing four or more fire events were within a distance of 200 to 300 m from canals and about 95% were between 200 m and 400 m, i.e. these burn scars were highly concentrated in the zone nearest to the canals. Consequently, close proximity to canals not only influences burned area depth but also the probability of high frequency fire events (i.e. four or more consecutive fires). The bias towards high frequency fires occurring closer to canals may explain the higher relative burned area depth for these fires.

For the multi-criteria analysis of burned area depth in relation to both fire frequency and distance to drainage canals, logarithmical trend lines were applied to each fire frequency class as functions of the distance to drainage canals (Fig. 5c). For all fire frequency classes relative burned area depth decreases with greater distance from canal. Concurrently, but operating independently of the distance to canals, the relative burned area depth decreases with every successive fire event. In consideration of the fact that the class of two fire events at large distances from drainage canals is described by a small number of sample points the assumed decrease in relative burned area depth for an increasing number of fire events can be approved. The difference between three and four to seven fire events, with five fire events on average for the higher frequency class, is 0.06 m. Hence, relative burned area depth from the fourth fire event onwards may be between >0 and 0.05 m with similar values per fire event. The multi-criteria analysis further rebuts the potential assumptions that higher burned area depth close to canals (Fig 5a) could be an exclusive result of higher fire frequencies in these areas (otherwise trend lines would be approximately constant) or, vice versa, higher burned area depth for higher fire frequencies (Fig. 5b) could exclusively result from closer

distances to canals (otherwise trend lines would approximately overlay). Both investigated factors affect burned area depth in an interdependent and mutually reinforcing way.

Validation of pre-fire peat surface subsidence measurements

As validation we compared our results with previous airborne LiDAR measurements, where an average depth of burn value of 0.33 m was determined for single fires predominantly in a zone of between 0 and 100 m from drainage canals (Ballhorn *et al.*, 2009). In the Ballhorn study, the fires investigated occurred shortly before the LiDAR data acquisition, hence other biological oxidation processes would not have played a significant role in the observed burned area depth, even with close proximity to canals. In our study, we observed a minimum value of 0.21 m for single fires at a distance of 200 to 300 m from canals. Assuming that subsidence decreases with distance to canals, our results can be considered to be consistent with those from Ballhorn *et al*. (2009).

Peat bulk density and carbon content values

The average bulk density of peat samples taken over the near-surface layer of 0.1–0.4 m depth in low drainage, unburnt, forested locations was 0.117 ± 0.018 g cm^{-3} (n=48) and 0.125 ± 0.017 g cm^{-3} (n=60) in high-drainage forest, yielding an overall average value of 0.121 ± 0.018 g cm^{-3} which was taken to be representative of areas experiencing a first fire. The near-surface peat from pits located in burnt areas had a somewhat lower average bulk density of 0.115 ± 0.020 g cm^{-3} (n=313) and this value was taken to be representative of all burnt areas, regardless of fire frequency. The average bulk density value for the forested sites is very similar to the average value of 0.122 ± 0.052 g cm^{-3} obtained by Warren *et al.* (2012) for the nearby forested Sebangau peatland.

The average carbon content of peat samples at 0.2 m depth was 55.3%, with a standard deviation of 4.2%. Average values for forested and burnt areas were identical. This is very close to the average value of $56.3 \pm 4.6\%$ reported by Jaya (2007) from the nearby Blok C and Blok E areas of deep peat;

also to the average value of 54.0 ± 3.3% reported by Warren et al. (2012) for the nearby Sebangau peatland; and to the average value of 56% determined by Page *et al.* (2011) for Southeast Asia as a whole.

Mass of peat fuel and carbon loss values

Using the above information on fire-related peat surface subsidence, combined with average values for bulk density and carbon content of peat in the study area, it is possible to provide average values for fire-driven peat fuel consumption and carbon loss (Table 3). In the absence of any specific information, a combustion efficiency of 1.0 was applied (a simplifying assumption which assumes complete combustion of all organic carbon) (Yokelson *et al.*, 1997). When no information about distance to canals is available, the values in Table 3 can be used for mass of peat fuel and carbon loss estimates for successive fires. We have simplified this to account for one, two, three and four or more fire events, on the basis that for fire frequencies over three, each fire results in a similar relative burned area depth of 0.02 m as a conservative assumption.

Where there is no information available on fire frequency, it would also be possible to apply the mass of peat fuel and carbon loss values presented in Table 4a which are based on the location of the burn scar in relation to distance from canal. Where information on both fire frequency and distance from canal is available, this allows greater stratification of peat surface subsidence and carbon loss (Table 4b). Alternatively, the trend functions applied to the single fire frequency classes could be used to calculate burned area depth for distances from drainage canals between 200 and 800 m (Table 4b). For larger distances, the constant values provided in Table 4 may be applied. For distances less than 200 m the values for the 200–300 m interval may be applied as a conservative approach.

Discussion

Our results for burned area depth are in the same order of magnitude as field measurements (Van Leeuwen *et al.*, 2014) and consistent with previous airborne LiDAR measurements published by Ballhorn et al. (2009), where a contour based approach was applied to derive average depth of burn directly from the LiDAR DTM. However, the advancement of the approach presented here is that through applying a spatially explicit model, the complex dependence of burned area depth on distance to canal and fire frequency can be mathematically described and the wide variation in depth within burned areas is better captured. Single measurements can attain burned area depths of up to -1.19 m and even for the same fire frequency at similar distances from drainage canals, standard deviations of 0.10 to 0.19 m for the means of the different classes are observed. By investigating the full extent of burned areas, a large number of sample points can be derived covering all combinations of fire frequencies and distance intervals within the study area. The sampling method further makes results less biased and more robust against possible errors due to uncertainties of both the LiDAR point cloud filtering and the interpolation methodology, with 95% confidence intervals being less than 0.015 m for all mean values.

With this novel approach we show a strong interdependence between burned area depth, distance to drainage canals and fire frequency. While the impact of drainage proximity on burned area depth is presumably a consequence of decreasing water table depths i.e. increasing moisture content of the upper peat column, with greater distance from canal, the decrease of burned area depth for successive fire events may be explained by a number of factors. These include the reduction in the amount of aboveground fuel load in terms of live and dead wood, which is mostly consumed by the first and second fires respectively. Concurrently, the change in post-fire vegetation cover, which is increasingly dominated by ferns and sedges (Fig. 2), leads to a faster speed of fire spread, with less chance of smoldering peat fires becoming established (Hoscilo *et al.*, 2011, 2013). Furthermore, chemical changes in peat during and after fires result in a reduction in the more labile, easily-

combustable constituents (Milner L, Boom A & Page SE. Effects of fire on the organic geochemistry of tropical peat. Manuscript in preparation).

In this study we also paid specific attention to the various factors that can result in a lowering of the peat surface in burned areas. The peatland in our study area has been strongly impacted by fire but also by other processes that contribute to lowering of the peat surface following drainage (i.e. physical compaction and biological oxidation). Since the entire area has been affected by drainage and has subsided, including the reference points, the modeled pre-fire surface does not represent the actual initial peat surface. Instead, the measurements used in our analyses have to be considered as *relative* measurements of the difference between burned and unburned peat surfaces. The non-fire processes that contribute to subsidence are most active shortly (up to five years) after drainage (Hooijer *et al.*, 2012). These processes had, therefore, already stabilized when the LiDAR data were acquired, enabling us to provide a clearer assessment of the specific role played by fire in peat and hence carbon loss. In addition, as shown in Hooijer *et al.* (2012), the lowered peat water table around drainage canals has a strong impact on all subsidence processes (biological oxidation, physical compaction, fire), bringing down the peat surface over distances of up to one kilometer from canals, even in the absence of fire (Hooijer *et al.*, 2014). In our study area, this impact was greatest in a zone of about 200 m from canals, i.e. the zone of greatest water table drawdown, and it was assumed that over larger distances, non-fire subsidence was more uniform than fire subsidence. By excluding this 200 m zone from both modelling and analyses, the relative effect of non-fire subsidence processes on measured subsidence was reduced.

With regard to the reporting of the mass of peat fuel combusted and associated carbon losses, we compared our results to the values provided by the IPCC. For the first fire, the Tier 1 IPCC methodology would have yielded a peat fuel consumption value of 353 ± 183 t ha^{-1} equivalent to a carbon loss of 195 ± 101 t C ha^{-1} (for a peat carbon content of 55.3% as applied to our results for first

fires). Using the revised values for burned area depth presented in this study, the peat fuel consumption value for a first fire is 206 t ha^{-1} with a carbon loss of 114 t C ha^{-1} (some 42% lower). The IPCC approach is based on depth of burn data from three studies, two of which acquired data during intense ENSO-related droughts (1997 and 2006) (Page *et al.*, 2002; Ballhorn *et al.*, 2009). Thus not only are our results more conservative but also a more accurate value to be used in further quantification and reporting of Indonesian peatland fire carbon losses across a more representative sample of fire years and not just those associated with extended ENSO-related dry seasons. For a second fire, the value for the mass of peat fuel is 115 t ha^{-1}, resulting in a carbon loss of 64 t C ha^{-1}; for a third fire these values are 69 t ha^{-1} and 38 t C ha^{-1}, respectively, and for fourth and subsequent fires the peat fuel and carbon loss values reduce to 23 t ha^{-1} and 13 t C ha^{-1}, respectively. In order to compare the scale of carbon losses using previous methodologies and the present revised approach, we calculated fire-related carbon loss across the whole study area shown in Fig. 1 and within the test sites. Based on the carbon loss values provided in Table 4b that involve a more detailed stratification, total carbon loss since 1990 amounts to 18.4 Mt C for the approx. 100,000 ha burnt area of the study area. When taking default values provided by the IPCC (Drösler *et al.*, 2014), however, carbon losses would have been overestimated by approximately 130% at 42.2 M t C. Within the test sites, values for total carbon loss are equally divergent with 876,000 t C using our revised approach and 2,115,000 t C (Drösler *et al.*, 2014). By way of comparison, carbon losses were calculated for the total volume of peat combusted (difference between the modelled surface and the LiDAR DTM), where the 200 m zone and logging tracks were excluded. They amount to 357,000 t C compared to 374,000 t C using the revised approach and 971,000 t C using the IPCC default value. Thus this revision of the methodology allows more accurate, stratified reporting of mass of peat fuel combusted and the associated carbon loss based on fire history. In combination with appropriate gaseous emissions factors (Drösler *et al.*, 2014) this should enable Indonesia to report its fire emissions from degraded peatlands at IPCC Tier 2 level.

Acknowledgements

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Supporting Information captions

Table S1 Burned area depth as a function of distance to drainage canals.

Table S2 Burned area depth as a function of fire frequency.

Tables

Table 1 Comparison of different interpolation techniques for modelling the pre-fire peat surface. Error statistics are related to the difference between modeled surface and Digital Terrain Model (DTM). They are based on 100,000 random points divided into two classes depending on their location in the test site. (IDW: Inverse Distance Weighted; NN: Natural Neighbour)

Reference area						
	Bézier	IDW	NN	Trend	Spline	Kriging
Mean [m]	0.000	0.000	0.001	0.000	0.001	0.001
Std [m]	0.134	0.138	0.145	0.303	0.225	0.147
Min [m]	-0.885	-0.777	-0.766	-3.579	-2.086	-0.773
Max [m]	0.614	0.731	0.722	0.982	2.729	0.704

Interpolated area						
	Bézier	IDW	NN	Trend	Spline	Kriging
Mean [m]	0.000	0.049	0.060	-0.071	0.878	0.070
Std [m]	0.133	0.187	0.138	0.168	9.780	0.208
Min [m]	-0.725	-0.822	-0.631	-0.863	-57.376	-0.935
Max [m]	0.454	0.805	0.504	0.495	112.834	0.862

Table 2 Results of Bézier approximation with different polynomial orders.

Polynomial order	5	**6**	**7**	**8**	**9**	**10**	11
Test site Block A/E							
RMSE [m]	0.143	0.133	0.131	0.128	0.128	0.127	0.331
Min [m]	-1.238	-1.159	-1.188	-1.187	-1.152	-1.131	-5.222
Max [m]	0.690	0.707	0.694	0.670	0.658	0.643	9.101
Test site Block B							
RMSE [m]	0.130	0.107	0.101	0.100	0.099	0.098	0.133
Min [m]	-1.323	-1.003	-0.866	-0.866	-0.853	-1.066	-2.183
Max [m]	0.964	0.618	0.587	0.580	0.570	0.595	1.959

Table 3 Values for mass of peat fuel and total carbon loss from first through to fourth+ fires (carbon

loss is reported *per fire event*).

Fire event	First fire	Second fire	Third fire	Fourth+ fire
Average relative burned area depth [m]	0.17	0.10	0.06	0.02
Mass of peat fuel (t ha^{-1})*	206	115	69	23
Carbon loss value [t C ha^{-1}]*	114	64	38	13

* Calculation of mass of peat fuel values assumes a peat bulk density value of 0.121 ± 0.018 g cm^{-3}

for the first fire (average for the upper 0.4 m of peat in forested areas in the study area; value is the

average of 0.117 + 0.018 g cm^{-3} in low-drainage forest (n=48) and 0.125 ± 0.017 g cm^{-3} (n=60) in high-

drainage forest) and of 0.115 ± 0.020 g cm^{-3} (n=313) for second and subsequent fires (being the

average for the upper 0.4 m of peat in burnt areas).Calculation of carbon loss values assumes a peat

carbon content of 55.3 ± 4.2% (average for peat at a depth of 0.2 m in the study area; this value

applies to both forested and burnt peatland; n=27); and a combustion efficiency of 1.0 (a simplifying

assumption based on complete combustion of all organic carbon).

Table 4 Values for total carbon loss from burned areas according to distance from drainage canal,

regardless of fire frequency (a) and considering fire frequency (b).

Distance to canal (DC) [m]		200–<300	300–<400	400–<500	500–<600	600–<700	700–<800	≥800
(a)								
Average burned area depth [m]		-0.38	-0.34	-0.28	-0.26	-0.23	-0.21	-0.19
Carbon loss value [t C ha^{-1}] *		254	228	187	174	154	141	127
(b)								
FF=1	Average burned area depth [m]	0.21	0.18	0.16	0.14	0.12	0.11	0.10
	Burned area depth [m] as function of DC (x): $f_1(x)=0.0875*\ln(x)-0.6905$, $x\in\Re$, $200\leq x<800$							
	Carbon loss value [t C ha^{-1}] *	**141**	**120**	**107**	**94**	**80**	**74**	**67**
FF=2	Average burned area depth [m]	0.37	0.31	0.26	0.23	0.20	0.17	0.15
	Burned area depth [m] as function of DC (x): $f_2(x)=0.1821*\ln(x)-1.3761$, $x\in\Re$, $200\leq x<800$							
	Carbon loss value [t C ha^{-1}] *	**235**	**197**	**165**	**146**	**127**	**108**	**95**
FF=3	Average burned area depth [m]	0.44	0.39	0.35	0.32	0.29	0.27	0.25
	Burned area depth [m] as function of DC (x): $f_3(x)=0.1590*\ln(x)-1.3220$, $x\in\Re$, $200\leq x<800$							
	Carbon loss value [t C ha^{-1}] *	**280**	**248**	**223**	**204**	**184**	**172**	**159**
FF=4+	Average burned area depth [m]	0.49	0.45	0.41	0.39	0.37	0.35	0.33
	Burned area depth [m] as function of DC (x): $f_{4+}(x)=0.1275*\ln(x)-1.1928$, $x\in\Re$, $200\leq x<800$							
	Carbon loss value [t C ha^{-1}] *	**312**	**286**	**261**	**248**	**235**	**223**	**210**

* Calculation of carbon loss values for (a) and (b) FF=1 assumes a peat bulk density value of 0.121 ± 0.018 g cm^{-3} for the first fire (average for the upper 0.4 m of peat in forested areas in the study area; value is the average of 0.117 + 0.018 g cm^{-3} in low-drainage forest (n=48) and 0.125 ± 0.017 g cm^{-3} (n=60) in high-drainage forest) and a peat carbon content of 55.3 ± 4.2% (n=27); and a combustion efficiency of 1.0 (a simplifying assumption based on complete combustion of all organic carbon). Calculation of carbon loss values for (b) FF=2–FF=4+ assumes a peat bulk density value of 0.115 ± 0.020 g cm^{-3} (n=313) for second and subsequent fires (being the average for the upper 0.4 m of peat in burnt areas); a peat carbon content of 55.3 ± 4.2% (average for peat at a depth of 0.2 m in the study area; this value applies to both forested and burnt peatland; n=27); and a combustion efficiency of 1.0 (a simplifying assumption based on complete combustion of all organic carbon).

Figure captions

Fig. 1 Fire frequency and drainage canals in the study area. The boxes indicate the test sites for the investigation of fire-related peat surface subsidence.

Fig. 2 Peatland in the study area that has been affected by one, two, three or more than three fires. Clearly visible is the decreasing amount of above-ground fuel potentially available for combustion: with each successive fire event the amount of deadwood decreases and vegetation becomes more heavily dominated by ferns and sedges.

Fig. 3 Distribution of NOAA and MODIS fire hotspots in relation to distance from drainage canals in the study area (a) and within the test sites (b).

Fig. 4 Exemplary LiDAR point cloud profile of 5 m width and 180 m length separated into ground an off-ground points. Different land cover types (peat swamp forest, burned area, drainage canal) are clearly visible. Dense low vegetation like ferns and swamp grasses lead to fewer ground returns – in

this example mainly covering the area burned once in 2009. The white rectangle superimposed on the ortho photo (bottom of the figure) shows the extent of this profile.

Fig. 5 Burned area depth in relation to fire frequency and distance to drainage canals. (a) Mean burned area depth as a function of the distance to canals by means of a line graph (without considering fire frequencies). (b) Mean burned area depth as a function of fire frequency (without considering distance to drainage canals). The cross indicates the mean cumulative burned area depth for each fire frequency class and the whiskers the standard deviation. The diagram shows that cumulative burned area depth increases with every subsequent fire event and that the relative burned area depth decreases for the first three fire events. (c) Burned area depth as a function of distance to drainage canals for each fire frequency (FF) class.

First fire

Second fire

Third fire

More than three fires

(a)

Frequency (%)

Number of hotspots

Distance to drainage canals (m)

(b)

Frequency (%)

Number of hotspots

Distance to drainage canals (m)

(a)

(b)

(c)