Carbon Emissions from Drained and Degraded Peatland in Indonesia and Emission Factors for Measurement, Reporting and Verification (MRV) of Peatland Greenhouse Gas Emissions

A summary of KFCP research results for practitioners

Al Hooijer, Sue Page, Peter Navratil, Ronald Vernimmen, Marnix van der Vat, Kevin Tansey, Kristina Konecny, Florian Siegert, Uwe Ballhorn, Nick Mawdsley

Kalimantan Forests and Climate Partnership
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Kalimantan Forests and Climate Partnership

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PREFACE

The Kalimantan Forests and Climate Partnership (KFCP) was a REDD+ demonstration activity established by the Indonesian and Australian governments and managed under the Indonesia-Australia Forest Carbon Partnership (IAFCP) along with the Indonesia National Carbon Accounting System (INCAS) Program.

KFCP was based on 120,000 hectares (ha) of degraded peatland in Central Kalimantan and operated from 2009 to 2014.

A key objective of KFCP was to generate new information and knowledge on peat swamp forests and degraded peatland as the basis for estimating peatland GHG emissions.

This working paper presents the key results and findings of the peatland GHG research conducted by KFCP and its implications for the measurement, reporting and verification (MRV) of GHG emissions from peatland, in particular, emission factors for MRV.

This work will be of interest to practitioners and technicians involved in GHG inventory and MRV, in particular those developing national and sub-national GHG MRV systems including the INCAS.

This working paper refers to a number of scientific publications which have been drafted and which will be published in international journals. A fuller account of the findings presented here will be available in these manuscripts when they become available.

This publication includes information and data that is being submitted to scientific journals. Users should check the most recent literature prior to using data in this report.
SUMMARY

The Kalimantan Forests and Climate Partnership (KFCP) program was set up to design and demonstrate methods to reduce carbon emissions from degraded peatlands and to quantify the effects of these interventions. Within KFCP, the studies reported here were set up to investigate the relations between land cover, water table depth, drainage intensity, subsidence rate, fire frequency and carbon emissions. A key target was to achieve robust GHG emission factors (emission factors) for degraded and burnt peatlands in Indonesia. This report presents a brief ‘summary for technicians and practitioners’ of the contents of three scientific papers that are in preparation, plus a translation of the work into emission factors.

The research was conducted over 2009-2014 by three collaborating research organizations: Deltares, Remote Sensing Solutions GmbH (RSS) and the University of Leicester. The study area is part of the Ex Mega Rice Project (EMRP) area in Central Kalimantan, Indonesia, where a peatland area of about more than half a million hectares was deforested, drained and largely burnt in 1995-1997 i.e. about 15 years before the study took place, without any agricultural or silvicultural use being developed subsequently.

In Section 2, ‘Annual carbon emissions from biological oxidation in 2011-12’, we report carbon emissions derived from monthly subsidence and water table measurements over the fire-free years of 2011-2012 at 431 locations. Average subsidence in burnt areas and forest is 0.87 cm yr⁻¹ and 1.53 cm yr⁻¹ respectively, with the highest rates occurring near canals. In intensively drained areas, subsidence is shown to be caused by peat oxidation and to be a direct measure of carbon loss. Relations between carbon emissions and water table depth are presented, both for the dominant land cover of previously burnt, degraded peatland and for the remaining pockets of forest. We calculate an average carbon loss from biological peat oxidation of 4.5 t C ha⁻¹ yr⁻¹ from burnt peatland and 7.9 t C ha⁻¹ yr⁻¹ from drained forest, excluding fire emissions. These Emission factors are consistent with those determined by CO₂ flux measurements in the same landscape, and with recent IPCC Emission factors. At the same water table depth, the emissions from degraded peatland are about half that from plantations and cropland indicating that management variables such as peat surface disturbance and fertilization may be as important in determining the peat oxidation rate as is water table depth.

In Section 3, ‘Cumulative carbon emissions since 1996 from both biological oxidation and fire’, we separate the processes involved in subsidence through the two impacts that it has on the peat: loss of surface elevation, as determined from airborne LiDAR transects, and changes to belowground bulk density as determined from soil samples along these transects. We find that in highly drained peatland, with 0.97 m of total subsidence in 15 years, oxidation through biological oxidation and fire explains 80% of the total loss of peat volume since drainage. We calculate that emissions over the first 5 years have been in the order of 80 t C ha⁻¹ yr⁻¹. The total amount of subsidence and peat carbon loss is found to have been 27% higher in burnt areas compared to areas that are drained but still forested. We conclude that the peat surface in the first few years following water table lowering through drainage re-adjusts to the new hydrological conditions regardless of whether fire occurs. The implication is that, in analyses of longer-term land subsidence and overall carbon loss, the fire component does not always have to be considered separately, but instead, two phases of carbon loss and subsidence after drainage may be distinguished: an initial brief phase of approximately five years with high but declining rates followed by a phase of lower but near-constant rates that in degraded peatlands will gradually decline as the peatland finds a new hydrological equilibrium (see Section 5).

In Section 4, ‘Carbon emissions from individual fire events’, we reconstruct pre-fire peat surfaces in burned areas based on airborne LiDAR data to determine the amount of fire-related peat subsidence as a function of fire frequency and distance to drainage canals. Based on the observed burn depths and the peat properties presented in the chapters above, Emission factors are presented for different numbers of fire events. It was observed that absolute peat subsidence increased and relative peat subsidence decreased with every fire
event. Hence, a clear correlation between the amount of peat subsidence in burned areas and fire frequency is evident. Average burn depth as a result of the first fire was observed to be 18 cm ± 2 cm, the second fire produced a burn depth of 11 cm ± 6 cm and third and subsequent fires 4.3 cm ± 2 cm. Resulting carbon emissions amounted to 120 t C ha⁻¹, 73 t C ha⁻¹ and 27 t C ha⁻¹ for the first, second and third and subsequent fires, respectively. A further finding is that the highest fire frequencies appear only close to canals, indicating that the impact of canal drainage not only influences the amount of peat subsidence, but also the probability of the re-occurrence of fires in these areas.

Section 5, on ‘Integration of studies: peatlands as self-organizing ecosystems and practical implications’, brings together observations on this topic, and discusses implications for peatland rehabilitation. We find that unless canal water levels continue to be lowered, as is the case in and around plantations (and croplands and roads), carbon loss will rapidly diminish in the first years after drainage regardless of whether forest regrowth or fires occur. There are several implications that should be considered in planning peatland use from a carbon emission reduction perspective. An implication is that carbon emissions from managed landscapes will always be much higher than from degraded but unmanaged peatlands, regardless of emission reduction measures such as managing plantation water levels at the highest possible level that is suitable for production. This is the case unless the management itself aims to bring up water levels close to the surface, as is ideally the case in peatland rehabilitation efforts.

Section 6 presents some practical ‘Recommendations for peatland MRV in Indonesia’, for GHG (CO₂) emissions both from biological oxidation and fires.

Section 7 presents ‘Peatland Emissions Factors for Indonesia’ resulting from KFCP studies in comparison with IPCC Emission Factors. It is first discussed that field measurements indicate a clear climate/temperature effect on the rate of peatland CO₂ emissions due to biological oxidation. Consequently, and inevitably, emissions from drained peatlands are higher in the tropics than elsewhere. For instance the IPCC CO₂ Emission factors for drained forest are about 10 times lower in boreal than in tropical areas, at 0.37-0.93 and 5.3 t ha⁻¹ yr⁻¹ respectively. Likewise, emission factors for croplands are about twice as high for tropical areas than temperate areas, at 14 and 7.9 t ha⁻¹ yr⁻¹, respectively.

Table 6 and Table 7 present the emission factors that the KFCP expert team recommends for estimation of emissions in national and sub-national carbon accounts in Indonesia, at the Tier 2 level. Figure 15 combines the biological oxidation and fire Emission factors in a way that should be easier to apply the values in the context of MRV procedures, accounting for the interrelations between the two emission types.
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<th>Description</th>
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<tbody>
<tr>
<td>DTM</td>
<td>Digital Terrain Model</td>
</tr>
<tr>
<td>EMRP</td>
<td>Ex Mega Rice Project</td>
</tr>
<tr>
<td>ENSO</td>
<td>El Nino Southern Oscillation</td>
</tr>
<tr>
<td>GHG</td>
<td>Greenhouse Gas</td>
</tr>
<tr>
<td>GWP</td>
<td>Global Warming Potential</td>
</tr>
<tr>
<td>Ha</td>
<td>Hectare</td>
</tr>
<tr>
<td>HD</td>
<td>High drainage</td>
</tr>
<tr>
<td>INCAS</td>
<td>Indonesia National Carbon Accounting System</td>
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<tr>
<td>IAFCP</td>
<td>Indonesia-Australia Forest Carbon Partnership</td>
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<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
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<tr>
<td>KFCP</td>
<td>Kalimantan Forests and Climate Partnership</td>
</tr>
<tr>
<td>Km</td>
<td>Kilometres</td>
</tr>
<tr>
<td>LiDAR</td>
<td>Light Detection and Ranging</td>
</tr>
<tr>
<td>m</td>
<td>Metre</td>
</tr>
<tr>
<td>MD</td>
<td>Moderate drainage</td>
</tr>
<tr>
<td>NEE</td>
<td>Net ecosystem exchange</td>
</tr>
<tr>
<td>MODIS</td>
<td>Moderate Resolution Imaging Spectroradiometer</td>
</tr>
<tr>
<td>MRV</td>
<td>Measurement, reporting and verification</td>
</tr>
<tr>
<td>NOAA</td>
<td>National Oceanic and Atmospheric Administration</td>
</tr>
<tr>
<td>RSS</td>
<td>Remote Sensing Solutions GmbH (RSS)</td>
</tr>
</tbody>
</table>
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1 INTRODUCTION

1.1 Background

Tropical peatlands in Southeast Asia have been deforested at record rates over the last ten years with recent trends indicating that nearly all peatland forest in Sumatra and Kalimantan in Indonesia could be lost by around 2020 (Miettinen et al. 2012a). Most of this peatland remains in a degraded state for a period of time before reverting to forest or being converted to plantations or cropland; the latter conversion usually takes at least five to ten years (Miettinen et al. 2012b, 2012c). Most of this unused and largely deforested peatland is drained, albeit not to the same degree as plantations and croplands in the same landscape. It has been shown that peatland deforestation and drainage cause carbon emissions, but that the emission rates vary with management conditions, including depth of the water table (Hooijer et al. 2010).

In recent years, an increasing number of studies have been published that provide carbon loss numbers for agricultural landscapes on peat, especially Acacia and oil palm plantations. Such studies are, however, still limited in number and scope for burnt peatland and degraded peat swamp forest that have not been converted to agriculture (collectively often referred to as ‘degraded peatlands’), despite the fact that such landscapes may be the most widespread land cover on Southeast Asian peatland. There is a need for improved insight into carbon emissions from such degraded areas and the impact of converting them to plantations, to provide a basis for regional emission assessments and improved land use planning.

Apart from carbon accounting, improved understanding of subsidence rates due to carbon loss in drained peatlands is also needed for long-term projections of future drainability and land use potential in these areas as land loss due to flooding is very likely in the coastal peatlands of Southeast Asia (Hooijer et al. 2012).

The Kalimantan Forests and Climate Partnership (KFCP) program was set up to design and demonstrate methods to reduce carbon emissions from degraded peatlands, and to quantify the effects of these methods. Within KFCP, the studies reported here were set up to investigate the relations between land cover, water table depth, drainage intensity, subsidence rate, fire frequency and carbon emissions. A key target was to achieve robust CO₂ emission factors for degraded and burnt peatlands in Indonesia. We consider this study to be a benchmark, as both the data availability and thoroughness in data collection and analysis are far greater than in most comparable studies.

The KFCP research was conducted over 2009-2014 by three collaborating research organizations, who led on different parts of the work and sections of this report: Deltares on oxidation emissions (Section 2) and the evolution of the landscape due to peat loss and subsidence (Section 3); and Remote Sensing Solutions GmbH (RSS) and the University of Leicester on fire-related emissions (Section 4).
1.2 The study area

The study area ( 
Figure 1) is part of the Ex Mega Rice Project (EMRP) area in Central Kalimantan, Indonesia, where a peatland area of about half a million (ha) was deforested, drained and largely burnt in 1995-1997 i.e. about 15 years before the study took place. The southern part of the study area of some 44,500 ha, called ‘Blok A North-West’, is delineated by the Kapuas and Mantangai rivers to the west, south and east, and by the major Main Primary Canal (*Saluran Primer Induk* or SPI) to the north. It is drained by a rectangular network of canals that are 2,600 metres (m) apart in a North-South direction, some 8 m wide and originally over 5 m deep. In 1997, drainage was followed, within months, by fires that destroyed most of the remaining forest cover (Page et al. 2002). Large-scale fires have since returned in dry years, either as a land clearing method that requires limited effort or as a consequence of various other causes with the result that most locations had burnt 2 to 4 times by 2012 but some up to 7 times (see Section 4). Burnt parts of the ‘Blok A’ area are now largely covered with low shrubs and ferns, although regrowth with fast-growing and relatively fire-resistant *Melaleuca cajuputi* (Gelam) trees occurs in some parts. A few pockets of degraded original forest remain, which have been affected by past logging and the drying effects of peat drainage.

The portion of the study area to the north of the main SPI canal is part of ‘Blok E’. It is about 75,000 ha in size, has been affected by the drainage from only one or two major canals and has retained a good forest cover despite industrial-scale concession-based selective logging in the 1980s and widespread small scale, selective, illegal logging in the following years up to this day. As a result of greater hydrological integrity and limited human activity there have only been a few fires of limited extent in Blok E (see Figure 2) and only near main canals. However, while Blok E mostly remains covered in closed canopy secondary forest with high biodiversity and conservation value, the logging has created a dense network of tracks and ditches and it must now be considered slightly but almost uniformly drained in its entirety. It may therefore be representative of most peat swamp forest that remains in Sumatra and Kalimantan, almost none of which is in pristine condition (Miettinen et al. 2012a).

The peat dome that was the target of the KFCP research is clearly visible in Figure 3.
1.3 This report

This report presents a brief ‘summary for technicians and practitioners’ of the contents of three scientific papers that present the outcomes of three interrelated but separate studies that used data collected in the KFCP project to determine different aspects of carbon emissions from drained and degraded peatlands in Indonesia. The outcomes of these studies are presented in the Sections 2, 3 and 4. To maintain the link with the KFCP scientific papers, these sections follow the standard for such publications (i.e. summary, followed by approach, methods, results and discussion).

Section 5 discusses how integration of the studies presented in the three preceding sections confirms that a more ‘holistic’ approach to peatland functioning and emissions from tropical peatlands is warranted, similar to views on the functioning of northern peatlands. Section 6 places the different types of emissions in a framework that allows them to be used for MRV and compares them with recent IPCC (2013) values. Section 7 presents some recommendations for a peatland GHG MRV system for Indonesia.
2 ANNUAL CARBON EMISSIONS FROM BIOLOGICAL OXIDATION IN 2011-12

This section presents the emissions related aspects of the scientific paper ‘Determining subsidence and carbon emission due to biological oxidation in degraded tropical peatlands 15 years after drainage, in relation to land cover and water table depth’ (Hooijer et al. 2014a, in preparation) to which we refer readers wanting to learn more about details and methods. We report carbon emissions derived from monthly subsidence and water table measurements over the fire-free years of 2011-2012 at 431 locations in an area of deep peat in Central Kalimantan that had been largely deforested, drained and burnt 15 years before without any agricultural or silvicultural use being developed subsequently. Average subsidence in burnt areas and forest less than 1,300 metres from canals is 0.87±0.55 cm yr⁻¹ and 1.53±0.70 cm yr⁻¹ respectively, with highest rates occurring near canals. In intensively drained areas, subsidence is shown to be caused by peat oxidation and to be a direct measure of carbon loss. Relations between carbon emissions and water table depth are presented, both for the dominant land cover of burnt peatland and for the remaining pockets of forest. We calculate an average carbon loss from biological peat oxidation of 4.5 t C ha⁻¹ yr⁻¹ from burnt peatland and 7.9 t C ha⁻¹ yr⁻¹ from drained forest; it should be noted that this excludes fire emissions. These emission factors are consistent with those determined by CO₂ flux measurements in the same landscape and with recent IPCC emission factors. At the same water table depth, the emissions from degraded peatland are about half that from plantations and cropland, indicating that management variables such as peat surface disturbance and fertilization may be as important in determining the peat oxidation rate as is water table depth.

2.1 Measurements and methods

Study locations were situated in all land cover types across the entire study area with the emphasis on the more diverse and accessible Blok A. Measurement locations for peat surface subsidence, water table depth, temperature and peat characteristics were organized along transects that were perpendicular to canal (Figure 2). To avoid the zone around canals that was potentially disturbed by heavy machinery during canal construction, no measurements were taken within a 30 m zone on either side of canals. Total transect length was 66 km. Field data were collected by two full-time field teams from early 2010 to January 2013.

Measuring subsidence rate and water table depth

Subsidence rates and water table depth were measured in dipwells made of perforated PVC tubes that were inserted at least 0.5 m into the mineral material below the peat, which was usually clay or loam, ensuring there would be no vertical movement. The distance from the top of the tube to a marker on the peat surface, as well as to the water table, was measured every month. The marker was made of a ring of 1 cm thick ‘ironwood’. Extensive precautions were taken to avoid disturbance during installation and monitoring of dipwells, and to remove disturbed records from the final analysis, as explained in Hooijer et al. (2014a, in preparation).

Calculating annual carbon loss

The method applied is described by Van den Akker et al. (2008; for temperate peatlands) and Couwenberg and Hooijer (2013; for Southeast Asian tropical peatlands), and with slight variations also applied by Gronlund et al. (2008) and Leifeld et al. (2011). This approach assumes, based on findings in earlier studies, that at some time after drainage, when peat compression processes (compaction and consolidation c.f. Andriesse, 1988) that initially contribute to land surface subsidence become negligible, 100% of remaining subsidence may be attributed to carbon loss due to oxidation. This is because the peat profile in the unsaturated zone above the water table is in a steady state with the rate of peat loss due to oxidation at the top being equalised by the supply of fresh peat entering from the saturated zone below as the depth of the unsaturated zone moves downwards over time, as illustrated graphically in Couwenberg and Hooijer (2013).
In this calculation the average bulk density of the peat below the lowest water table was applied in carbon loss calculations, i.e. a bulk density of 0.089±0.014 g cm$^{-3}$ (as measured from samples taken over the depth range 1.6–2.4 m below the peat surface in Blok A), whereas a carbon concentration value of 58% (Hooijer et al. 2014a, in preparation) was used for the peat below the water table.

**Table 1: Statistics for water table depth, subsidence and carbon loss in the study area**

<table>
<thead>
<tr>
<th>Drainage Condition</th>
<th>Study Area</th>
<th>Land Cover Condition</th>
<th>Number of Locations</th>
<th>Peat Thickness (m)</th>
<th>Mean WTD (m)</th>
<th>Maximum WTD (m)</th>
<th>25 percentile WTD (m)</th>
<th>Subsidence rate (cm yr$^{-1}$)</th>
<th>Peat bulk density 1.6-2.4 m (g cm$^{-3}$)*</th>
<th>Carbon concentration at 0.6 m (%)*</th>
<th>Carbon loss (t ha$^{-1}$ yr$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Drained by large canals</td>
<td>Blok A</td>
<td>Burnt</td>
<td>220</td>
<td>6.92</td>
<td>-26</td>
<td>-0.63</td>
<td>-0.37</td>
<td>0.87</td>
<td>0.089</td>
<td>0.58</td>
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<tr>
<td></td>
<td>Lightly drained</td>
<td>Blok E</td>
<td>Degraded forest</td>
<td>62</td>
<td>1.57</td>
<td>0.13</td>
<td>0.17</td>
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<td>0.014</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Forest</td>
<td>107</td>
<td>7.42</td>
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<td>0.86</td>
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<td></td>
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<td>0.68</td>
<td>0.17</td>
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<td>1.24</td>
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</table>

*From Hooijer et al. (2014a, in preparation)*

2.2 Results

**Measured water table depth patterns and fluctuations**

Water levels were highest on average in the burnt and degraded peatland, with an average depth of 0.26±0.13 m below the surface, and lowest in the degraded and drained forest in Blok A, at 0.43±0.17 m (Table 1). The lightly drained forest in Blok E has intermediate average water table depth, at 0.34±0.18 m. Locations near canals generally had greater average water table depths than those further away, as shown in Figure 3. At all locations, the water table showed seasonal fluctuations, which were usually about 0.50 m in both 2011 and 2012 (Figure 3). It should be noted that these were two relatively wet years, and that water table fluctuations as well as average water table depth would probably be somewhat greater in most years.
Subsidence rate

At most locations, the peat surface showed seasonal fluctuations of several centimetres, following fluctuations in water level, albeit with much smaller amplitude. Surface levels dropped around 4 cm on average in the dry season to then rise again by around 2 to 3 cm going into the wet season (Figure 3). Subsidence rates were determined by calculating the change in surface levels in the wet seasons over the two year period, from January 2011 to January 2013.

The seasonal variation in peat surface position observed in this study is also commonly observed in studies of temperate peatlands (Schothorst 1977; Whittington et al. 2007; Fritz et al. 2008) and is attributed to shrinkage and swelling of the peat above the water table, as moisture content varies with rainfall rates. This pattern has not been described before in drained tropical peatlands, for reasons explained in Hooijer et al. (2014a, in preparation).

The average subsidence rate was lowest on average in the burnt peatland, at 0.87±0.55 cm yr⁻¹, and highest in the most intact but slightly drained forest in Blok E, at 1.99±1.24 cm yr⁻¹ (Table 1). The degraded forest in Blok A has an intermediate subsidence rate, at 1.53±0.70 cm yr⁻¹.

Carbon loss as calculated from subsidence over 2011–2012

Average carbon loss in burnt peatland and degraded forest in Blok A is 4.5 and 7.9 t C ha⁻¹ yr⁻¹ respectively. For the slightly drained forest in Blok E, it was not possible to calculate carbon loss from subsidence measurements because here the physical processes of consolidation and compaction appear to still have a substantial effect, as explained in Hooijer et al. (2014a, in preparation).
Carbon loss in relation to water table depth

The relation between carbon loss and water table depth, determined separately for the different land cover types and calculation methods, is as follows (Figure 4; see Hooijer et al. 2014a, in preparation, for details):

For burnt peatland:

\[ CL = -9.32\times WD + 2.12 \]  \[1\]

For degraded forest:

\[ CL = -12.01\times WD + 2.80 \]  \[2\]

Where:

\( CL \) = carbon loss (includes CO\(_2\), DOC and CH\(_4\) emission) [t C ha\(^{-1}\) yr\(^{-1}\)]

\( WD \) = average water table depth [m; negative if below the surface].

Carbon loss in relation to distance to canal

For assessments of peatland carbon emission where no accurate long-term measurements of water table depth and subsidence are available, as is often the case, it can be helpful to express carbon loss as a function of distance to canal, which then serves as a measure of drainage impact. The following relation for carbon loss is derived by applying Equations 1 and 2 to water table depth as a function of distance to canal (Figure 5):

For burnt peatland:

\[ CL = -0.93\times \ln(DC) + 9.57 \]  \[3\]

For degraded forest:

\[ CL = -1.84\times \ln(DC) + 19.06 \]  \[4\]

Where:

\( CL \) = carbon loss (includes CO\(_2\), DOC and CH\(_4\) emission) [t C ha\(^{-1}\) yr\(^{-1}\)]
DC = distance to canal [m]

It is advised that these relations should only be applied to distances of more than 30 m from canals, which is the minimum distance at which subsidence measurements were taken in this study. For smaller distances from canals, the values at 30 m may conservatively be applied. It should also be noted that these equations were developed for, and only applicable to, large canals that are usually specifically constructed for drainage or boat transport (i.e. not for ditches meant for timber log transport); such large canals will typically have a width of greater than 5 m.

Figure 5: Carbon loss rate (calculated from subsidence rate) as a function of distance to canal and land cover type, in the drained Blok A area

2.3 Discussion

Comparing carbon loss as determined from subsidence to CO₂ flux measurements, for degraded tropical peatland

a) Burnt peatland

The average carbon loss in burnt peatland as determined by the subsidence method, of 4.5 t C ha⁻¹ yr⁻¹, is close to the NEE (Net Ecosystem Exchange) value of 5.0 t C ha⁻¹ yr⁻¹ measured by a flux tower over an 8 year period as reported by Hirano et al. (2012) for a peatland area in Blok C approximately 50 km away that has similar land cover and fire history (}
Figure 6). Moreover, from parallel measurements by 6 automated flux chambers, Hirano et al. (2013) find net heterotrophic respiration in the same burnt peatland area in Blok C to be 3.8 and 3.6 t C ha\(^{-1}\) yr\(^{-1}\) for two measurement periods of 2004-2005 and 2005-2006 respectively. The latter values exclude fluvial carbon loss, which is in the order of 1 t C ha\(^{-1}\) yr\(^{-1}\) for degraded peatland in Central Kalimantan (Moore et al. 2013), which brings the total carbon loss closer to the values obtained using the subsidence and flux tower methods.

The similarity of the carbon emission numbers for burnt peatlands found through three very different approaches – subsidence measurements, flux towers and flux chambers – confirms the validity of each of the approaches if applied over sufficiently long periods and with sufficiently large numbers of measurements. Each of these methods does not strictly represent only the emissions from heterotrophic respiration for a number of reasons. Firstly, CO\(_2\) emissions as determined from subsidence may still include some carbon loss through DOC (fluvial carbon loss) and CH\(_4\) (methane) emissions. Second, flux tower NEE results are affected by net carbon uptake if biomass is increasing, although this does not occur as rapidly in burnt peatlands as it does in degraded forest. Third, flux chamber measurements can include autotrophic respiration from roots. Yet the relatively small differences in the results yielded by the three methods suggest that these factors may be almost negligible in a degraded, tropical peatland environment where CO\(_2\) emissions from peat oxidation are the dominant carbon flux.

b) Drained peatland

The average carbon loss in drained, forested peatland as determined by the subsidence method is 7.9 t C ha\(^{-1}\) yr\(^{-1}\). The NEE values reported by Hirano et al. (2012), from flux tower measurements in Blok C of slightly drained and degraded peatland forest respectively, of 1.7 and 3.3 t C ha\(^{-1}\) yr\(^{-1}\), are well below the values found in this study. This may be explained by the vigorous carbon uptake in regenerating tropical forest, which dominates the carbon balance and therefore the NEE measurement in that environment, thereby yielding a much lower net emission value. The IPCC (2006) provides a default Tier 1 value of 6.1 t C ha\(^{-1}\) yr\(^{-1}\) for the annual aboveground net biomass growth of tropical forests less than 20 years of age in insular Asia and 1.6 t C ha\(^{-1}\) yr\(^{-1}\) for tropical forests more than 20 years of age.

**Oxidation carbon loss from burnt and degraded peatlands as compared to cropland and plantations in Southeast Asia**

As shown in
Figure 6, emissions from cropland and plantations (Acacia and oil palm) on tropical peatland have been intensively studied in recent years, and are within a fairly narrow range if water table depth is accounted for. Only limited differences are apparent between the results of flux and subsidence methods in most studies. It is evident that carbon emissions from burnt peatlands and degraded forests are about three or four times below emissions from peatlands converted to plantations or cropland. This is in line with the IPCC (2014) numbers for these tropical land use types of 5.3 t C ha\(^{-1}\) yr\(^{-1}\) for drained forest land and cleared forest land and ~15 t C ha\(^{-1}\) yr\(^{-1}\) for drained plantations and can be explained by at least two factors. Firstly, actual water table depth in plantations is usually at least double that in degraded peatlands. Second, even at the same water table depth, degraded peatlands have about half the carbon loss of plantations or croplands reflecting different land management regimes. Further reasons for this large difference are explained in Hooijer et al. (2014a, in preparation).

**Implications for emission reduction in degraded and burnt peatlands**

The relation reported here between water table depth and carbon loss (Figure 6) confirms that bringing up water levels in the study area, by blocking canals, will reduce carbon emissions. The benefits of such an intervention are likely to be evident first near canals, where water table depths and emissions prior to intervention are greatest. The water table lowering effect of drainage is less severe in areas further away from canals, certainly some years after the implementation of drainage when the peat surface has largely readjusted to the lower water tables through subsidence: thus in these locations the short-term benefits of canal blocking for reduction of carbon emission due to peat oxidation are more limited.

It was found that water levels and peat stability in all locations in the study area were affected by drainage, including the Blok E area that was not drained by major canals but by a multitude of logging ditches that were previously thought to have limited impact. This illustrates the need to prevent any drainage in peatlands, including logging ditches, if carbon emissions are to be avoided. If peatlands are used for timber production, harvesting and extraction methods that do not require drainage, such as light railway systems, are required to minimize carbon loss.
Figure 6: Comparison of relations between average water table depth and CO2 emission (or carbon loss, in the case of subsidence studies) as determined in the study area and other peatlands in Southeast Asia

NB: Studies in forest, burnt peatland, oil palm plantations and other types of cropland and plantations are indicated by green, orange, red and purple lines and symbols respectively. The results of subsidence studies are shown as thick solid lines. Results of flux measurements are not corrected for autotrophic (‘root’) respiration except for Jauhiainen et al. (2012) who reduced total soil emission values by 12% to determine actual heterotrophic respiration caused by peat oxidation, and Hirano et al. (2013) who chose sites with no or little vegetation growth in order to exclude autotrophic respiration.
3 CUMULATIVE CARBON EMISSIONS SINCE 1996 FROM BIOLOGICAL OXIDATION AND FIRE

This section presents the highlights of the scientific paper ‘Separating carbon emissions from oxidation and fire in drained tropical peatland using surface morphology, subsidence rates and bulk density profiles’ (Hooijer et al. 2014b, in preparation) to which we refer readers wanting to learn more about details and methods. We separate the processes involved in subsidence through the two impacts that it has on the peat: loss of surface elevation, as determined from LiDAR transects, and changes to belowground bulk density as determined from soil samples along these transects. We find that in highly drained peatland at around 50 m from canals, with 0.97 m of total subsidence in 15 years, oxidation (through biological oxidation and fire) explains 80% of the total loss of peat volume since drainage, while compaction (i.e. compression above the water table) explains 20%. Total carbon loss over these 15 years is ~463 t C ha\(^{-1}\). We calculate that emissions over the first 5 years have been in the order of 80 t C ha\(^{-1}\) yr\(^{-1}\). The total amount of subsidence and peat carbon loss over 15 years is found to have been 27% higher in burnt areas compared to areas that are drained but still forested. We conclude that the peat surface, in the first few years following water table lowering through drainage, re-adjusts to the new hydrological conditions regardless of whether fire occurs. The implication is that in analyses of longer-term land subsidence and overall carbon loss, the fire component does not have to be considered separately, but that instead two phases of carbon loss and subsidence after drainage may be distinguished: an initial brief phase of approximately 5 years with high but declining rates followed by a phase of lower but near-constant rates that however will fluctuate with weather conditions.

3.1 Approach

One of the most important consequences of the drainage of peatlands is land subsidence resulting from the progressive loss of peat volume from the landscape. In drained peatlands, the oxidative loss of peat carbon is known to be the main contributor to the lowering of peat surface elevation, although other physical processes are also involved. Hooijer et al. (2012) summarized the processes of most importance in tropical conditions as: 1) peat oxidation by biological (microbial) decomposition, 2) peat oxidation by fire, 3) peat compression by compaction and shrinkage of the top layer of peat above the water table, and 4) peat compression by consolidation of the entire peat column. All of these processes contribute to subsidence, but they operate in different ways and over varying timescales, and only the first two processes result in carbon loss. All processes are most active near canals, where the drainage effect is greatest (Figure 7).
Figure 7: Schematic illustration of the two main groups of processes, compression and oxidation that contribute to subsidence following drainage

NB: this concept applies in areas where canals are > 1,000 m apart, not to plantations where drainage density is such that nearly uniform water table depths and carbon loss rates occur.

In the KFCP project area, we determined the cumulative carbon emissions arising since drainage as a result of both biological decomposition and fires with a focus on understanding emissions in the initial few years after drainage. A secondary objective was to separate the contributions from biological oxidation and fire to total emissions.

3.2 Measurements and methods

Measurements took place along two parallel transects of 14 km length and 4.6 km apart, which ran southward into Blok A from the SPI canal (Figure 2). Each transect traversed all types of land cover, drainage conditions and fire history to be fully representative of the site. To minimize the effect of natural variation in peat characteristics, peat samples for bulk density measurement were taken along the shortest possible stretches of the transects that still crossed all land cover types, which was a length of 3.4 km. All peat sampling locations were near dipwells where water table depth and surface subsidence rates were also measured over 2010-2013 as reported by Hooijer et al. (2014b, in preparation).

Sampling location grouping

Peat sampling locations were grouped by land cover, fire history and distance to canal. Going from least to most disturbed conditions, the following data ‘Clusters’ were defined (Figure 8):

- **Forest with Least Drainage** (F_LD) – Forest with the least drainage impact and no fire disturbance. Locations are between 1000 and 1250 m from the nearest canal. This cluster serves as the reference for relatively undisturbed natural conditions.

- **Forest with High Drainage** (F_HD) – Forest with high drainage impact but no fire disturbance. Locations are approximately 50 m from canals (locations closer to canals are avoided because there may have been soil disturbance by excavators during canal digging).

- **Burnt peatland with Moderate Drainage** (B_MD) – Burnt peatland with relatively moderate drainage impact. Locations are 100 to 400 m from canals.

- **Burnt peatland with High Drainage** (B_HD) – Burnt peatland with high drainage impact. Locations are approximately 50 m from canals.
**Peat profile sampling and bulk density analysis**

Peat samples for analysis of bulk density and other parameters were taken from 1 m² pits that were dug to depths of up to 2.5 m into the peat during dry periods. All peat samples were dried at 105 °C at multiple time steps of 24 hours. The bulk density analysis was considered final when dry weight had stabilized with less than 1% weight loss difference between subsequent measurements, which generally happened within 72 hours. A total of 1437 samples from 29 soil pits were available for this assessment.

**Elevation data**

Airborne LiDAR data were collected for the full KFCP area in the dry season of 2011, with a point density of 2 to 8 points m⁻². To generate a Digital Terrain Model (DTM) from these data, the effect of vegetation was removed by selecting the minimum values over a 25*25 m grid. The 25*25 m resolution is a compromise between larger grid cells (>50*50 m) that may provide higher accuracy in flat areas, but do not allow accurate detection of relatively steep surface slopes near canals, and small grid cells (<10*10 m) that may be more suitable for detection of slopes but are insufficiently able to filter out all vegetation signals.

**Estimating original peat surface levels**

A surface elevation level and slope for 2011 along study transects was determined from the current elevation of soil pits at 1,000–1,250 m from canals in the forested and least drained (F_LD) areas. From this surface connecting the highest points in the profile, a surface slope was determined that is thought to be parallel to the peat surface before intensive drainage started i.e. in 1996 (Figure 8). The elevation of the reference level was determined from the elevation more than 1 km to the north of the SPI canals where drainage has been least intensive. This yields a reference level that is 0.53 m higher than current surface elevation at the F_LD cluster locations, referred to as REF_1996_PROB. This level is in accordance with subsidence rates 15 years after drainage, of 1.29 cm yr⁻¹ at locations 1000 m from canals (0.19 m in 15 years) at F_LD locations, plus the knowledge that these subsidence rates must have decreased over the years (Stephens and Speirs, 1969; Andriesse 1988; Deverel and Leighton, 2010; Hooijer et al., 2012).

Figure 8: Profiles along the two transects showing peat surface elevation as affected by drainage since 1996, historical fire frequency, and generalized sampling locations
The two transects are shown in Table 2. Also shown here are the reference levels for estimated 1996 surface elevation levels.

**Calculating the contribution of oxidation and compression to total subsidence, from bulk density profiles**

For each peat sampling location, the current surface level was determined as well as the difference with the reference level approximating the original surface elevation. The differences in surface elevation were then averaged over the Clusters to suppress random variations (resulting from spatial patterns in local peat characteristics and water table depths). Profiles of bulk density were also averaged over Clusters.

For Cluster averages, the relative contributions from oxidative processes (biological decomposition and fire) and compression processes (consolidation and compaction) are then calculated in a number of steps that are explained in Hooijer et al. 2014b, in preparation.

### 3.3 Results

**Bulk density profiles**

In all profiles bulk density in the top peat is consistently higher than in the underlying peat, with average values of 0.118±0.017 g cm⁻³ (mean ± SD) over the top 1.5 m and 0.089±0.014 g cm⁻³ below it respectively (Table 1). The SD values are consistently around 15% of average values at all depths within each Cluster, showing that lateral variations in natural peat characteristics and recent carbon loss dynamics are within a limited range. The differences in bulk density between Clusters are also limited, varying from 0.115±0.020 g cm⁻³ for burnt, moderately drained sites (B_MD) to 0.124±0.018 g cm⁻³ for forested sites with high drainage intensity (F_HD) over the top 1.5 m and, for the peat below that depth, to between 0.088±0.013 g cm⁻³ (for F_HD and B_MD sites combined) and 0.092±0.021 g cm⁻³ for forested sites with least drainage (F_LD).

Calculation of emission factors values assumes a peat bulk density value of 0.121 g cm⁻³ (average for the upper 0.4 m of peat in forested areas in the KFCP area; value is the average of 0.117 g cm⁻³ in low-drainage forest and 0.125 g cm⁻³ in high-drainage forest (see Section 2); a peat carbon content of 55% (average for peat at a depth of 0.2 m in the KFCP area; this value applies to both forested and burnt peatland (see Section 2); and a combustion efficiency of 1.0 (a simplifying assumption based on complete combustion of all organic C).

**Surface subsidence as determined from LiDAR**

Relative to the REF_1996_PROB reference level, the B_HD, F_HD and B_MD (burnt and forested high drainage and burnt moderate drainage) site clusters are estimated to have subsided by 1.08±0.32, 0.86 ±0.12 and 0.79±0.14 m respectively since drainage in the mid-1990s.

**Peat and carbon loss caused by oxidation (biological oxidation and fire)**

The probable total carbon loss over 1996-2011, i.e. below the reference level, varies in highly drained peatland from 390 t C ha⁻¹ for F_HD (forested high drainage) to 536 t C ha⁻¹ for B_HD (burnt high drainage) with a difference between the two of 27% and an average of 463 t ha⁻¹. From subsidence and flux monitoring we know the annual loss over 2011-2012 to be 7.9 and 4.5 t C ha⁻¹ yr⁻¹ in burnt peatland and drained forest respectively (Hooijer et al. 2014a, in preparation; Hirano et al. 2012; Hirano et al. 2013) and we may assume that this rate has applied from 5 years after drainage onwards if not longer (Hooijer et al., 2012). Accounting for this lower rate of carbon loss after 5 or less years, the minimum loss rate over the first 5 years after drainage becomes 311 and 491 t C ha⁻¹ for F_HD, and B_HD respectively, or 62 and 98 t C ha⁻¹ yr⁻¹ on an annual basis with an average of 80 t C ha⁻¹ yr⁻¹.
3.4 Discussion

Relevance to carbon accounting

The findings of this study can for certain purposes greatly simplify the understanding of carbon loss in drained and burnt tropical peatlands, as it is not always necessary to consider the effects of biological decomposition and fire separately. For such applications the emission numbers presented here may serve as default carbon emission values for highly drained tropical peatlands regardless of fire history. For other applications, especially if there is a need to quantify specific fire emissions i.e. GHG (CO₂, CH₄), a more complex approach would still be required as offered by the parallel fire investigations in this study as reported by Konecny et al. 2014 (in preparation).
The significance of the emission spike in the first 5 years in highly drained peatland

The results of this study confirm that carbon emissions in the first few years after drainage, certainly in highly drained areas, are far higher than in subsequent years, regardless of the occurrence of fire. The average annual emission from highly drained peatland of 80 t C ha\(^{-1}\) yr\(^{-1}\) or 294 t CO\(_2\) ha\(^{-1}\) yr\(^{-1}\) in the initial 5 years after drainage is of the same order as the value of 178 t CO\(_2\) ha\(^{-1}\) yr\(^{-1}\) for Acacia plantations provided by Hooijer et al. (2012) for the same period and using a similar approach, in a landscape that was not affected by fire.

It appears that plantation-style drainage of any tropical peatland, at least if the peat is fibric-hemic and with a low mineral concentration typical of ombrogenous domed peatland, causes an initial emission spike in the order of 200 t CO\(_2\) ha\(^{-1}\) yr\(^{-1}\) averaged over the first 5 years after drainage, regardless of land cover change and of whether fire takes place. In reality, most of this emission spike is likely to occur during an even briefer period of maybe less than one or two years directly following drainage when labile surface peat is rapidly decomposed (Hooijer et al. 2012; IPCC, 2013) or burnt (Page et al. 2002), but this cannot be determined with the data available. Spreading this peak emission over five years accounts for the possibility that drainage systems may be developed in steps over several years, as is sometimes the case in smaller scale plantations, but this does not affect the results of carbon emission calculation results over longer periods. The actual duration of this peak may become clearer in future.

As indicated in recent publications (Hooijer et al. 2012; IPCC, 2013), this initial emission spike is now not accounted for in emission calculations for specific crops grown on tropical peatlands, or in calculations of regional emissions for Southeast Asia (Hooijer et al. 2010). The data presented here confirm that a correction of this underestimation is necessary and possible.
4 CARBON EMISSIONS FROM INDIVIDUAL FIRE EVENTS

This section presents the findings of the study presented in the paper ‘Fire related peat subsidence and carbon loss in drained tropical peatlands in Central Kalimantan, Indonesia’ (Konecny et al. 2014, in preparation). This study employed a new methodology for reconstructing pre-fire peat surfaces in burned areas based on airborne LiDAR data to determine the amount of fire-related peat subsidence as a function of fire frequency and distance to drainage canals. Based on the observed burn depths and the peat properties presented in the sections above, emission factors are presented for different numbers of fire events. The highest subsidence in burned areas was found in the close proximity of canals, however, a significant portion of the subsidence in this zone must be attributed to non-fire subsidence processes. Consequently, a zone of 200 m along canals was excluded from calculation of average burn depths. It was observed that absolute peat subsidence increased and relative peat subsidence decreased with every fire event, even when taking into account the spatial variability due to varying drainage intensity. Hence, a clear correlation between the amount of peat subsidence in burned areas and fire frequency is evident. Average burn depth by first fires was observed to be 18 cm ± 2 cm, second fires produced a burn depth of 11 cm ± 6 cm and third and subsequent fires 4.3 cm ± 2 cm. Resulting carbon emissions amounted to 120 t C ha⁻¹, 73 t C ha⁻¹ and 27 t C ha⁻¹ for the first, second and third and subsequent fires, respectively. A further finding is that the highest fire frequencies appear only close to canals, indicating that the impact of canal drainage not only influences the amount of peat subsidence, but also the probability of the re-occurrence of fires in these areas.

4.1 Measurements and methods

LiDAR Data filtering and DTM generation

In order to generate a DTM that serves as the basis for the quantification of peat subsidence, filtering of the LiDAR data to distinguish between ground and off-ground surfaces was undertaken. A methodology similar to that described in Pfeifer et al. (2001) was used. Selection of the lowest points over a 5 m grid was then made. The sampling grid size of 5 m was used to ensure that we could better capture the surface characteristics of relatively small burn scars and improve the DTM where dense vegetation would otherwise preclude the detection of the ground elevation by LiDAR. A geo-statistical method (Kriging) was used to create the surface from the LiDAR data (Kraus 1998). For each land cover class, a detailed accuracy assessment was carried out. The 5 m resolution DTM proved to have a vertical accuracy of 0.12 m for peat swamp forest (when compared to differential GPS height measurements) and was used for the further analyses.

Delineation of burned areas relating to fire frequency

In a first step, MODIS and NOAA hotspots were used to identify years of fire occurrence in the study area (1990, 1997, 2001, 2002, 2004, 2005, 2006, 2007, 2008, 2009 and 2011). Burned areas were delineated based on historical Landsat and recent RapidEye imagery. Fire frequency was determined by intersecting the burned areas for every year of fire occurrence (Figure 10).
Identification of drainage canals and logging trails

Drainage canals and recent logging tracks (skid trails and small ditches) in the KFCP area were identified by visual interpretation of the LiDAR DTM. The location of drainage canals and logging ditches was required in order to be able to exclude areas directly adjacent to canals and ditches from the analysis and to assess the impact of drainage depth, as a function of distance to canals, on fire-related peat subsidence.

Spatial modelling of pre-fire peat surface

In order to reconstruct the pre-fire peat surface, selected burned areas were spatially interpolated based on reference elevation values of adjacent unburned areas. Suitable burned areas had to meet the conditions of a) being situated completely on peat, and b) having an unburned reference area on every side (Figure 11).
For the surface reconstruction a Bézier approximation was performed for each test site. The reference points were situated in unburned areas that had a distance of at least 200 m from drainage canals and 30 m from burned areas.

The difference between the modelled reconstructed peat surface and the LiDAR-derived DTM indicates a measure of peat subsidence after fire at each pixel and was used for the quantification and analysis of spatial variability in burned areas due to various factors of influence.

**Quantification of peat subsidence due to fire considering fire frequency and distance to drainage canals**

For the quantification of peat subsidence due to fire a sample-based method was applied. Approximately 50 randomly placed points per ha were generated within the burned area, excluding a buffer zone of 200 m either side of drainage canals and 20 m on either side of small logging ditches. Additionally a buffer zone of 30 m around the total extent of the burned area was excluded to avoid edge effects. The datasets were intersected both with spatial (DTM, modeled surface, normalized modeled surface, distance to drainage canals) and thematic (fire frequency, year of fire occurrence) information.
4.2 Results

Peat subsidence in burned areas in relation to fire frequency

The relative difference between the means of consecutive classes decreases up to a fire frequency of three (i.e. subsidence for the first to third fires is 0.177 m, 0.112 m and 0.043 m, respectively, Table 2, Figure 12) and remains stable for the fourth fire (0.044 m). The increase between the classes representing four and five fire events results from the fact that 82% of the observed points with a fire frequency of five or more are primarily located close to a canal, where we have observed greater amounts of subsidence due to non-fire processes.

Table 2: Peat subsidence as related to fire frequency

<table>
<thead>
<tr>
<th>Fire frequency</th>
<th>Cumulative peat subsidence (cumulative peat burn depth) [m]</th>
<th>Peat subsidence (peat burn depth) of individual fire events [m]</th>
<th>Number of points</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.18</td>
<td>0.177</td>
<td>20035</td>
</tr>
<tr>
<td>2</td>
<td>0.29</td>
<td>0.112</td>
<td>2918</td>
</tr>
<tr>
<td>3</td>
<td>0.33</td>
<td>0.043</td>
<td>29664</td>
</tr>
<tr>
<td>4</td>
<td>0.38</td>
<td>0.044</td>
<td>5137</td>
</tr>
<tr>
<td>5</td>
<td>0.54</td>
<td>0.16</td>
<td>5514</td>
</tr>
<tr>
<td>6+</td>
<td>0.51</td>
<td>+0.03</td>
<td>3571</td>
</tr>
</tbody>
</table>

Figure 12: Relationship of peat subsidence (m) (peat burn depth) to fire frequency for all measurement points > 200 m from a drainage canal
**Peat subsidence in burned areas in relation to distance to drainage canals**

A significant relationship between distance to drainage canal and peat subsidence as a function of fire frequency was observed (Pearson correlation coefficient of $r=0.860$ calculated using mean peat subsidence values and midpoints of distance classes). Excluding a zone of 200m around each main canal produced a stronger relationship ($r=0.929$). Peat subsidence decreases logarithmically with greater distance from canals with an impact of the canals observed up to a distance of at least 600 m. The mean peat subsidence in burned areas, i.e. the mean peat burn depth, is -0.309 m, and for distances greater than 600 m from canals it is 0.217 m (Table 3, Figure 13).

**Table 3: Peat subsidence as a function of distance from canal**

<table>
<thead>
<tr>
<th>Distance to canal [m]</th>
<th>Peat subsidence (i.e. peat burn depth) [m]</th>
<th>Number of points</th>
</tr>
</thead>
<tbody>
<tr>
<td>200-&lt;300</td>
<td>0.38</td>
<td>27836</td>
</tr>
<tr>
<td>300-&lt;400</td>
<td>0.35</td>
<td>19727</td>
</tr>
<tr>
<td>400-&lt;500</td>
<td>0.29</td>
<td>12041</td>
</tr>
<tr>
<td>500-&lt;600</td>
<td>0.27</td>
<td>8723</td>
</tr>
<tr>
<td>600-&lt;700</td>
<td>0.24</td>
<td>6326</td>
</tr>
<tr>
<td>700-&lt;800</td>
<td>0.22</td>
<td>5076</td>
</tr>
<tr>
<td>800-&lt;900</td>
<td>0.22</td>
<td>3758</td>
</tr>
<tr>
<td>900-&lt;1000</td>
<td>0.21</td>
<td>3067</td>
</tr>
<tr>
<td>1000-&lt;1100</td>
<td>0.20</td>
<td>2302</td>
</tr>
<tr>
<td>1100-&lt;1200</td>
<td>0.20</td>
<td>1779</td>
</tr>
<tr>
<td>1200-&lt;1300</td>
<td>0.19</td>
<td>1056</td>
</tr>
<tr>
<td>&gt;=1300</td>
<td>0.17</td>
<td>876</td>
</tr>
</tbody>
</table>

**Peat subsidence in burned areas as a function of fire frequency and distance to drainage canals**

The examination of the distribution of fire frequency classes related to their distance to canals shows that areas with a large number of fire events are found close to canals. Hence, the distance to drainage canals not only influences the amount of peat subsidence, but also the number of fire events in the study area (Figure 13).
Fire frequency classes were summarized into three classes (1, 2-3, >4). In all three classes, the peat burn depth decreased with greater distance from drainage canal. Concurrently, but operating independently of the distance to canal, peat subsidence (i.e. the cumulative peat burn depth) increased with every additional fire event (Table 4;
In the peatland within 200 m of the canals, peat subsidence is strongly influenced by drainage-induced biological oxidation and compaction, which increases with increasing proximity to canals. Therefore, this zone was excluded from our analysis.

Table 4: Peat subsidence measured in zones with increasing distance from canals as a function of fire frequency

<table>
<thead>
<tr>
<th>Fire Frequency</th>
<th>Peat subsidence/Number of points</th>
<th>Distance from Canal (m)</th>
<th>Average</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>200-&lt;300</td>
<td>300-&lt;400</td>
</tr>
<tr>
<td>1</td>
<td>Peat subsidence [m]</td>
<td>-0.21</td>
<td>-0.19</td>
</tr>
<tr>
<td></td>
<td>Number of points</td>
<td>5210</td>
<td>4231</td>
</tr>
<tr>
<td>2-3</td>
<td>Peat subsidence [m]</td>
<td>-0.43</td>
<td>-0.39</td>
</tr>
<tr>
<td></td>
<td>Number of points</td>
<td>5704</td>
<td>5164</td>
</tr>
<tr>
<td>4-7</td>
<td>Peat subsidence [m]</td>
<td>-0.51</td>
<td>-0.46</td>
</tr>
<tr>
<td></td>
<td>Number of points</td>
<td>8066</td>
<td>4672</td>
</tr>
<tr>
<td>Average</td>
<td></td>
<td>-0.40</td>
<td>-0.35</td>
</tr>
</tbody>
</table>
Fire emission factors

Using the above information on depth of burn for first and subsequent fires, peat bulk density and carbon content values (provided earlier in Section 3) it is possible to provide average fire emission factors for the land covers present in the KFCP study area (Table 5).

Table 5: Emission factors for total carbon loss from first through to third fires in the KFCP area (carbon loss is reported per fire event)

<table>
<thead>
<tr>
<th>Fire event</th>
<th>First fire</th>
<th>Second fire</th>
<th>Third+ fire</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average depth of burn (m)</td>
<td>0.18</td>
<td>0.11</td>
<td>0.04</td>
</tr>
<tr>
<td>Emission factor (t C ha⁻¹)*</td>
<td>120</td>
<td>73</td>
<td>27</td>
</tr>
</tbody>
</table>

* Calculation of emission factor values assumes a peat bulk density value of 0.121 g cm⁻³ (average for the upper 0.4 m of peat in forested areas in the KFCP area; value is the average of 0.117 g cm⁻³ in low-drainage forest and 0.125 g cm⁻³ in high-drainage forest; see Section 2); a peat carbon content of 55% (average for peat at a depth of 0.2 m in the KFCP area; this value applies to both forested and burnt peatland; and a combustion efficiency of 1.0 (a simplifying assumption based on complete combustion of all organic C).
4.3 Discussion

The results of this study demonstrate that absolute peat subsidence increased and relative peat subsidence decreased with every fire event, even when taking spatial variability due to varying drainage intensity into account. Hence, a clear correlation between peat subsidence and fire frequency can be observed. In addition, there is a relationship between the amount of peat subsidence in burned areas and the distance to canals, such that peat subsidence decreases with increasing distance from these drainage features. While biological oxidation will make an important contribution to the rate of peat subsidence that occurs close to canals, where the intensity of drainage is greatest (e.g. up to 200 m from canals) some share of the subsidence in this zone must also be attributed to fire, especially when two or more fires have occurred. However, as it is not possible to separate fire-related subsidence and subsidence due to biological oxidation and compaction, this zone was excluded here. A further finding of this study is that the higher fire frequencies appear only near canals, indicating that canal drainage not only influences the amount of peat subsidence, but also the probability of the re-occurrence of fires in these areas. The proportions of peat subsidence that can be attributed mainly to fire or to other subsidence processes will depend on various factors including the past disturbance of the ecosystem (including drainage intensity, and disturbance of vegetation cover). Since drainage in the study area had already commenced some 13 years before the start of this study and the processes that contribute to subsidence are most active shortly (5 years) after drainage (Hooijer et al. 2014b, in preparation) it is only possible to make inferences on the scale of carbon losses that may have been attributable to fire or other processes causing subsidence during the earliest phase of peatland drainage in the KFCP area. These inferences are underpinned due to the fact that the LiDAR data was acquired at a time when the drainage-induced subsidence processes had already stabilized, thus enabling a clearer definition of the role played by fire in peat, and hence, carbon loss.
5 INTEGRATION OF STUDIES: PEATLANDS AS SELF-ORGANIZING ECOSYSTEMS AND PRACTICAL IMPLICATIONS

Self-adjustment of drained peat systems

The landscape of the Blok A study area, which in its natural state had a surface slope of less than 0.5 m km\(^{-1}\), has been strongly transformed by drainage (Figures 1, 7, 8). Differential subsidence has created new surface slopes around canals that now often exceed 2 m km\(^{-1}\), and an egg-box shaped landscape of ‘mini domes’ has evolved in which the canals, rather than previous natural drainage patterns, are now the controlling landscape features.

Accounting for the simultaneous changes in peat surface elevation and peat bulk density 15 years after intensive drainage, we have shown that the peat system within a few years largely readjusts to the new hydrological conditions, regardless of whether fire has occurred, but also that full readjustments continue 15 years after drainage and probably for decades to come, in smaller and smaller annual increments. In areas that have not burnt, the adjustment is found to be somewhat slower, but the end result is very similar. This self-regulation capacity of peatlands, resulting in a relatively rapid readjustment of the peat surface to lower water tables, was also described for northern peatlands (e.g. Ingram, 1982; Price, 2003) and is discussed for Southeast Asian peatlands by Dommain et al. (2010) albeit in that case for intact, forested peatlands.

It appears that in areas that are not burnt soon after drainage, biological decomposition will in a few years cause amounts of carbon loss and subsidence that are nearly as high as those resulting more rapidly from fire. The two processes are different pathways by which oxidation processes result in the peat landscape readjusting to the new hydrological conditions. In both cases, this landscape adjustment results in a great reduction in water table depth, i.e. the deep water table conditions immediately after drainage are returned to near-natural water depths conditions within 15 years, and probably most of this readjustment occurs within 5 years (Hooijer et al. 2012). This is evident from water table records, as discussed in Hooijer et al., (2014a, in preparation) that show the water table regime 15 years after drainage to be not much different from the natural water table regime of an undrained peat dome. From Figure 7 it can be understood that the water table depth near canals in the study area has been drawn down by several metres immediately after drainage, causing highest subsidence rates where water depths were greatest, and resulting in a peat surface shape that now, some 15 years after drainage, approaches the water surface shape created by the initial drainage condition.

However it should be noted that this is not yet a stable situation; while water table depths and fluctuations are now far closer to ‘natural’ conditions than they were shortly after drainage, subsidence and emissions are still ongoing albeit at lower rates. Without canal blocking intervention these impacts would most likely continue for many decades to come, especially as the canal bottoms and therefore the drainage base in the KFCP area are seen to be lowered through erosion. Raising water levels in canals will set a higher and stable drainage base, allowing carbon loss and subsidence to end sooner than would be the case without intervention.

Implication for carbon loss and accounting

In general analyses of land subsidence and carbon loss, the results of this study indicate that the fire component does not have to be considered separately but, instead, two phases of carbon loss and subsidence after drainage should be distinguished: an initial brief phase of high but declining rates followed by a phase of lower but constant rates. For specific analysis where the timing and type of carbon emission is important, and where the assumption that most emission occurs as CO\(_2\) is not sufficient, fires may be accounted for separately as is possible on the basis of the findings of the separate study reported in Section
4 and in Konecny et al. (2014, in preparation). This would be necessary where a methodology required separate accounting of different carbon gases which have very different GWP values (e.g. CO₂ and CH₄).

**Implications for peatland rehabilitation**

The consequence of the peat surface adjustment shown in this study is that, unless canal water levels continue to be lowered as is the case in and around managed peatlands (plantations, croplands and roads), carbon loss will rapidly diminish in the first years after drainage regardless of whether forest regrowth occurs or fires occur. There are several implications that arise from this that should be considered in planning of peatland use from a carbon emission reduction perspective. The first is that carbon emissions from managed landscapes will always be much higher than from degraded but unmanaged peatlands, regardless of emission reduction measures such as managing plantation water levels with best practice water management that is suitable for production. This is the case unless the management itself aims to bring up water levels close to the surface, as is ideally the case in rehabilitation efforts.

Another implication is that subsidence rates and emissions in drained but abandoned peatlands will fall over time even without water management rehabilitation interventions (i.e. canal blocking), as the peat system readjusts to the new hydrological situation. At the same time, our finding of continued carbon loss after 15 years shows that the degree to which the landscape surface was disturbed is such that: i) this hydrological readjustment will probably take decades, regardless of water management interventions and ii) the effect of water management interventions on carbon emissions may largely be incremental rather than immediate. It should also be noted, however, that rehabilitation efforts usually also aim to allow (somewhat) natural forest to be restored, through fire prevention coupled to tree planting schemes with local communities, which bring additional benefits beyond the effect of water management interventions (Page et al., 2009). Therefore, our findings support the idea that peatland rehabilitation should apply a range of ecosystem restoration measures, rather than focusing only on emission reduction interventions that are likely to have only relatively limited impact in the short term.

The overall impact of water management rehabilitation interventions (i.e. canal blocking) in terms of reduced carbon loss should be measured over the long-term by comparing: i) the amount of carbon likely to be lost from the system without raised canal water levels to the future point in time when the hydrological readjustment process has been completed and a new equilibrium is established (i.e. the Reference Emission Level) to ii) the actual carbon lost over time. This will require an assessment and continuous monitoring of the impact of raising canal water levels on peatland water table depths, subsidence and carbon loss across the whole mini-dome peatland landscape through pilot canal blocking interventions linked to monitoring.
6  RECOMMENDATIONS FOR PEATLAND MRV IN INDONESIA

6.1  Key KFCP findings for carbon accounting

Several KFCP research findings presented in this paper are likely to have a substantial impact on carbon accounting science for tropical peatlands and on peatland policy discussions in Indonesia:

1. The KFCP research confirms that, in well-drained areas (the Blok A part of the KFCP area), nearly all subsidence since drainage is caused by the process of peat oxidation (whether that is brought about by biological (microbial) decomposition or fire) with the compression contribution being negligible beyond the first few years. This further simplifies the use of subsidence as a robust measure of carbon emission under such conditions, and should lead to even wider adoption of this method in emission monitoring. Subsidence monitoring is proposed as a key approach for Indonesia’s national MRV strategy for peatland GHG emissions as part of a continuous process of verification and improvement of emission factors.

2. It also finds, however, that subsidence in forest that is only slightly drained (the Blok E part of the KFCP area) is still strongly influenced by compression processes and, under these specific conditions, subsidence is not a good measure of emissions. As direct flux measurements are also problematic in such areas due to the dominance of the vegetation carbon cycle (when using flux towers) and CO₂ emissions from root respiration (when using flux chambers) there is still no robust method to quantify emissions from forest such as in Blok E that has good canopy cover and no major canals but much historical disturbance from logging and small ditches (’tatas’). As this type of forest is probably most typical for the majority of remaining forested Sumatran and Kalimantan peatlands that are not yet fully degraded or converted, and is also the main land cover in most planned peatland REDD projects, this presents a major challenge for carbon accounting and crediting.

3. We report strong evidence that emissions in the initial few years (up to 5 years) after drainage far exceed those in subsequent years. This was noted in earlier publications, and the latest IPCC (2013) emission factors explicitly exclude the first years after drainage for that reason. In previous studies, emissions during the immediate post-drainage period could not be quantified with the degree of accuracy and confidence that is now possible. If accounted for, inclusion of these initial emissions may well double average emissions from plantations over a 30 year period after drainage, which will greatly change the recognized emissions associated with crops grown on peat, such as oil palm and Acacia.

4. These initial emissions occur whether the peatland burns or not, as the peatland surface is found to readjust to the post-drainage water table as a result of oxidation processes, be they biological or fire-induced. In terms of carbon accounting this can greatly simplify methodologies over longer periods, removing the need to account for individual fires, but only when total carbon loss is being considered. This approach does not provide information on the quantities of GHGs or GWP which are quite different (e.g. CO₂ vs CH₄).

5. Emissions from individual fires are also quantified and two key findings were identified. Firstly, it was found that emissions from fire strongly decrease with every repeat fire, with emissions from the third to seventh fires being only a fraction of those from the first or second. The second key finding is that the fire emissions decrease with increasing distance from canals due to the gradient in water table depth. The fire Emission factors proposed in this report may be applied where it is necessary to consider separately the emissions from fires alone, such as in assessments of fire-related air pollution and of emissions under different fire management scenarios beyond the initial phase after drainage (in situations when the presence of fires does not need to be accounted for separately in calculation of the total carbon emissions) and also where the GWP is of relevance.
6.2 Field monitoring of emissions from biological oxidation

Brief comparison of methods

Three different approaches are applied in determining carbon emissions from drained peatlands: subsidence, the flux chamber method and eddy covariance. Measurements of land subsidence have the benefits of: i) integrating the impact of variable conditions over long time periods, ii) covering total carbon loss arising through different processes, iii) being relatively cost effective and simple, iv) allowing large numbers of locations to be monitored simultaneously, and v) being fully replicable and verifiable as the soil record remains in place (Hooijer et al. 2012). Methods to separate the oxidation (carbon loss) contribution to subsidence from that of physical compression have been demonstrated in many studies including Stephens and Speir (1969), Wösten et al. (1997), Deverel and Leighton (2010), Hooijer et al. (2012) and Couwenberg and Hooijer (2013), yielding values from 60% to over 90% for cumulative subsidence since drainage, including the early stage when compression is dominant. In Europe, an oxidation percentage of 70% is usually applied based on studies in the 1970s (Kasimir-Klemedson et al. 1997), but in later stages, after drainage, this approaches 100% especially in warm climates (Van den Akker 2008; Leifeld et al. 2011; Couwenberg and Hooijer 2012). The disadvantages of the subsidence approach are that at least two years of measurement time are required (although longer records are more representative in variable climates, so two years coverage is often considered a minimum for thorough research regardless of the method used) and that no separation of emission types (i.e. gaseous vs. aquatic fluxes) is possible.

Direct measurement of carbon (CO₂ and CH₄) fluxes from the soil using chamber methods, on the other hand, can require less time, but allows only limited measurement numbers and locations and there is no verification record. Gaseous carbon exchanges measured using this method have revealed large spatial and temporal variability (e.g. Bubier et al. 2003) resulting in potentially high uncertainty when scaling up (Page et al. 2011). In part, these issues can be overcome by using the eddy covariance or ‘flux tower’ method which, in contrast to closed chambers, is able to provide continuous, multi-year, whole ecosystem gaseous flux measurements over relatively large (ha to km²) areas. An increasing number of studies in northern peatlands have used this method, but there are few published eddy covariance studies addressing CO₂ balances from tropical peatlands (Hirano et al. 2012). Peatland emissions as presented by the IPCC 2013 Guidelines are determined using all three methods, which show good agreement if executed well (see also
Application of the subsidence method

The subsidence method was applied in the KFCP project as it was considered to be the most suitable to conditions in Indonesia (flux chamber measurements were also planned originally, at a smaller scale, but did not take place due to constraints on budget, time and field staff capacity). The high spatial variability in water table depth and subsidence rates in the KFCP study area illustrates the need to have a monitoring system that includes a sufficient number of measurement points and is well distributed throughout the study area. Similarly, the seasonal fluctuations in water table depth and peat surface position necessitate measurements over 2 years at least, and more if the number of monitoring locations is relatively small. With such a comprehensive and long-term system in place, the quantification of carbon loss from degraded tropical peatlands by measuring subsidence rates yields accurate emission numbers, at least for areas that are drained by major canals. However, the method is not recommended for lightly drained forested peatland, where it cannot be excluded that subsidence may be partly caused by peat compaction and consolidation rather than oxidation.

Using the relatively low-tech subsidence method brings benefits in terms of transparency and cost, and can provide a robust monitoring tool for carbon loss in drained peatland, if certain conditions in terms of drainage intensity and measurement scale and duration are met.

This confirms that to obtain a meaningful subsidence record it is necessary, apart from having sufficiently large numbers of measurement locations to cover spatial variability, to measure subsidence over a period of at least 2 years even if a large number of locations are monitored with high accuracy, as in this study. If fewer locations are monitored than in this study, as is usually the case, the minimum record would probably need to be substantially longer than 2 years. Records of more than 5 years are recommended for practical application beyond research projects that have the resources for large monitoring systems and tight quality control.

6.3 National monitoring of emissions from biological oxidation

When quantifying carbon emissions from tropical drained peatlands, it should be considered that these emissions are highly variable in space and time. Controlling factors are time since drainage, water table depth (a function of distance to canal and canal management), other management conditions (fertilization, soil disturbance), soil temperature (which is largely controlled by canopy cover), fire history and peat type. While conditions and emissions inside plantations may often be considered more or less uniform, substantial variations exist between plantations and even more so in unmanaged deforested areas. This is particularly evident in areas where undrained peatland is located adjacent to intensively drained croplands or plantations. In such situations, the drainage impact will extend into the supposedly undrained peatlands over distances of several kilometres in the longer term, as the effect of subsidence on hydrology progresses over decades (Hooijer et al. 2012). Roads and logging canals in forested peatland can have similar effects.

Ideally, these spatial and temporal variations in emissions should be accounted for. This would require ‘Tier 3’ level quantification of emissions, using models and field measurements. At present, the science base and monitoring system to allow a nationwide Tier 3 MRV system for peatland emissions is not available (although it would be possible for the EMRP area based on the KFCP results in Blok A in combination with the eddy covariance systems set up in Blok C and Sebangau NP (run by the Science and Technology Research Partnership for Sustainable Development project, funded by JICA and the Japan Science and Technology Agency (JST).
6.4 Field monitoring of emissions from fire

The parameters required to accurately calculate carbon emissions from peat fires are: a) the extent of the burned area; b) the depth of peat burn (to calculate the mass of peat fuel); c) the peat bulk density and carbon content; and d) the combustion efficiency of the fire. If numbers for specific gaseous products of combustion are required, then suitable emission factors (e.g. for CO$_2$ and CH$_4$) can be applied.

Up until the current study there was very limited information on the average depth of burn for different land covers and drainage conditions and no studies describing the changes in peat fuel consumption that occur as a result of recurrent fires. The methodologies used in this study to assess both burnt area and depth of burn follow similar approaches used in earlier studies of fire-driven carbon losses from tropical peatland landscapes (i.e. Page et al. 2002; Ballhorn et al. 2009) which used a combination of optical remote sensing, GIS, interpolation and LiDAR.

Although the total number of studies on depth of burn in tropical peats is still very small, in combination the results indicate a trend of reducing depth of burn with increasing fire frequency and reducing aboveground fuel load, which itself is related to site fire history. This conclusion is also consistent with the new information reported here which indicates a declining burn depth with each repeat fire in the same location. This reduction in depth of burn is likely a consequence of both reducing fire fuel load (i.e. replacement of high biomass forest by low biomass scrub or fern-dominated vegetation following the second and subsequent fires) but also a change in the physical and chemical characteristics of the peat substrate, which after each fire contains an increasing amount of recalcitrant soil organic matter that is more resistant to pyrolysis in subsequent fires (Milner L., S. Page and A Boom, unpublished data). It is also worth noting that the extent of fire-driven ecosystem modification will not only influence the severity of subsequent fires but also influence the fire return interval, operating again through effects on the quantity and quality of both above- and belowground fuel loads (Hoscilo et al. 2013).
7 PEATLAND EMISSION FACTORS FOR INDONESIA

The results of the KFCP peatland research have been used to update the peatland emission factors presented by IPCC (2013). The emission factors presented here are valid at the Tier 2 level; they are specific to Indonesia but do assume uniform emissions over large areas.

7.1 Summary of IPCC findings, placing tropical peatland emissions in a global context

The latest insights from studies of emissions from drained peatlands are reflected in the 2013 update of the IPCC Guidelines. The emission factor values for biological oxidation exclude all CO₂ emissions in the first 5 years after drainage, as well as emissions from fires and potential emissions from particulate organic carbon (POC) flushed into aquatic ecosystems.

The field measurements that are now available indicate a clear climate/temperature effect on the rate of peatland CO₂ emissions, especially for peatlands under well drained land uses which have the highest emissions. Consequently, and inevitably, emissions from drained peatlands are higher in the tropics than elsewhere. For instance the IPCC CO₂ emission factors for drained forest are about 10 times lower in boreal than in tropical areas, at 0.37-0.93 and 5.3 t ha⁻¹ yr⁻¹ respectively, while temperate emissions are about half at 2.5 t ha⁻¹ yr⁻¹. Likewise, CO₂ emission factors for croplands are about twice as high for tropical areas than temperate areas, at 14 and 7.9 t ha⁻¹ yr⁻¹, respectively.

It should be noted that in recent years, it has also become clear that drainage of peatlands not only produces high emissions of CO₂ from the land surface, but also of CH₄ from ditches and canals, where peat detritus flushed from the land accumulates and decomposes under anaerobic conditions. For grasslands on peat soils in temperate climates, the IPCC 2013 Guidelines propose a ‘ditch’ CH₄ emission of 527–1,156 kg ha⁻¹ yr⁻¹, which translates to a CO₂-equivalent emission of 18-39 t CO₂eq ha⁻¹ yr⁻¹ using a 100-year GWP factor of 34. The ditch CH₄ emissions from tropical plantations were found to be even higher, at 2,259 kg ha⁻¹ yr⁻¹ or 77 t CO₂eq ha⁻¹ yr⁻¹. Such high ditch CO₂eq emissions even exceed the actual known land CO₂ emissions from these areas on a per ha basis.

The state of knowledge on carbon emissions from fires on organic soils is not as well developed as that for emissions arising from biological oxidation. This is true of fires on organic soils in all regions of the world, not just the tropics. Up until the research undertaken as part of the KFCP study and described earlier in this report, there had only been a very small number of studies reporting the mass of peat consumed during fires on tropical peatlands. In addition, the effect of fire frequency on the scale of fire-derived carbon emissions from peatlands had not been demonstrated. The default data for organic soil fuel consumption during tropical peatland fires as published in the recent IPCC Wetlands Guidelines (IPCC 2013) are 353 ± 183 t ha⁻¹ (reported as mass of dry fuel) for wildfires on drained peatland (based on the results of three studies; equivalent to 235 ± 122 t C ha⁻¹ using values for peat bulk density of 0.121 g cm⁻³ and C content of 55% as applied in this report) and 155 ± 73 t ha⁻¹ for prescribed fires used during agricultural land management (based on the results of two studies; an emission of 103 ± 49 t C ha⁻¹). As a result of the research presented earlier in this report, we suggest that the default EF for initial (first) fires in the KFCP area should be 120 t C ha⁻¹. This is towards the bottom end of the IPCC range for wildfires on drained peatland (i.e. 235 ± 122 t C ha⁻¹). The lower values obtained for the KFCP study area can be explained by the fact that the IPCC emissions factor is influenced by the high fuel mass value obtained by Page et al (2002) for the 1997/98 fires. These fires occurred during a combined ENSO-IOD (El Nino Southern Oscillation – Indian Ocean Dipole) event that led to extreme lowering of the peatland water table and desiccation of above ground fuels, which were abundant as a result of extensive land clearing and canal construction during the initiation of the MRP. These conditions may therefore be somewhat atypical for initial fires on drained tropical peatland. In comparison, the mass of fuel data for first fires in the KFCP area cover a range of fire events across a number of years, not
all of which were associated with ENSO events. Therefore, we think that the fire emission factors resulting from the KFCP work are more representative of normal fire conditions than the emission factors now presented in the IPCC Guidelines, and we assume that during a next iteration of these guidelines, the results of this study will serve to correct the IPCC emission factors.

The second notable feature of the KFCP fire emission data is that, for the first time, it has been possible to demonstrate the diminishing mass of peat fuel available for combustion during subsequent fire events. We therefore not only present Emission factors for first fires, but also for second and up to fourth fires, thus enabling a more conservative approach to emissions reporting for peatlands that are subject to recurring fires. The improved knowledge will support reporting of fire Emission factors at the Tier 2 level.

7.2 Biological oxidation emission factors for MRV

Table 6 and Table 7 present the emission factors that the KFCP expert team recommends for estimation of emissions in national and sub-national carbon accounts in Indonesia, at the Tier 2 level.

Black numbers in Table 6 are derived from the KFCP research as presented in the earlier sections of this report. Green numbers are expert judgment. Blue numbers are derived from the IPCC 2013 guidelines. Purple numbers are derived from peer-reviewed literature. Red numbers are derived from the values in adjoining classes, or otherwise by estimating values from other classes, and must be considered to be uncertain.

Following IPCC practice, we have estimated emissions for land cover classes for which neither the KFCP research nor the IPCC 2013 guidelines present emission factors, by averaging values for ‘adjoining’ classes for which emission factors were available, i.e. classes that were considered to have somewhat lower and higher emissions respectively.

For instance, the emission more than 5 years after drainage for ‘slightly drained peatland forest’ (Class B; selectively logged but no large canals within 1000 m and not burnt, such as Blok E in the KFCP study area), was estimated by averaging the emission factors for intact forest (Class A; EF ‘0’) and moderately drained forest (Class C; EF ‘7.9”), yielding an emission factor value of 3.95. This same value was then applied to emissions over the first 5 years, as we have no information to calculate this otherwise.

The emissions over the first 5 years in Classes C and D (‘moderately drained’ and ‘fully degraded’ peatlands, respectively) were also estimated, from Class B (slightly drained forest) and Class E (plantation). These initial peak emissions in plantations were earlier found to be 178 t CO₂ ha⁻¹ yr⁻¹ or 49 t C ha⁻¹ yr⁻¹ (for Acacia plantations in Riau; Hooijer et al. 2012). The KFCP study found a somewhat higher peak emission of 80 t C ha⁻¹ yr⁻¹ at locations 50 m from canals, but this is of course not representative for the wider landscape further away from canals, where drainage impact and emission are less. For the purpose of carbon accounting for the wider landscape, we conservatively averaged the plantation emission of 49 t C ha⁻¹ yr⁻¹ with the Class B (slightly drained forest) value of 3.95 t C ha⁻¹ yr⁻¹, yielding a combined value of 26 t C ha⁻¹ yr⁻¹ for Classes C and D.
### Table 6: Emission Factors for Biological Oxidation on Indonesian peatlands at the Tier 2 level

<table>
<thead>
<tr>
<th>Land Cover Type</th>
<th>Emission Factor Biological Oxidation</th>
<th>&gt;5 years after drainage (only biological oxidation)</th>
<th>First 5 years after drainage (oxidation and/or fire)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>t C ha(^{-1}) yr(^{-1})</td>
<td>t C ha(^{-1}) yr(^{-1})</td>
</tr>
<tr>
<td>A. Primary peatland forest, never logged and without drainage impacts [now extremely rare in Sumatran and Kalimantan peatlands]</td>
<td></td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>B. Slightly drained peatland forest (selectively logged but no large canals within 1500 m and not burnt) [such as Blok E in the KFCP study area]</td>
<td></td>
<td>3.95</td>
<td>3.95</td>
</tr>
<tr>
<td>C. Moderately drained peatland forest (selectively logged and drained by large canals at 1000-3000 m intervals, but not burnt) [such as forested parts of Blok A in the KFCP study area]</td>
<td></td>
<td>7.9 [5.3; IPCC]</td>
<td>26</td>
</tr>
<tr>
<td>D. Fully degraded peatland (usually burnt at least twice with limited regrowth) that is drained by large canals at 1000-3000 m intervals [such as part of Blok A in the KFCP study area]</td>
<td></td>
<td>4.5 [5.3; IPCC]</td>
<td>26</td>
</tr>
<tr>
<td>E. Plantations and croplands, with large canals at intervals of less than 1 km and/or field drains less than 400 m apart [not found in the KFCP study area, but well covered by other studies]</td>
<td></td>
<td>15; IPCC</td>
<td>49 [Hooijer et al. 2012]</td>
</tr>
</tbody>
</table>
7.3 Fire emission factors for MRV

The emission factors for peat fires are reported per fire event as t C ha\(^{-1}\). Fires are distinct events, occurring in some years but not in others, and fire emission factor values cannot therefore be annualized (i.e. cannot be reported as t C ha\(^{-1}\) yr\(^{-1}\)). This follows the IPCC (2013) Guidelines. We therefore present the emission factors for biological oxidation separate from those for fire but using the same Land Cover categories. In preparing Table 7 we have:

a) Simplified the land cover types presented in Table 6 (land cover types B and C are combined as we have insufficient data at this stage to support derivation of separate fire emission factors).

b) Assumed that fires will not occur in land cover type A (based on previous studies which demonstrate that intact peat swamp forest is at very low risk of fire).

c) Based on the definition of land cover type D, assumed that one fire will already have occurred (of a magnitude equal to that of a first fire in land cover types B and C) and that emission factors are therefore only reported for second or subsequent fires.

d) Assumed that the first fire in land cover types B and C will occur during the first 5 years following drainage, and most likely during the first 2 years. The emission could therefore be assigned equally to the first 5 years (conservative) or across the first 2 years (less conservative).

The fire emission factors for land cover types B, C and D are based on field data obtained during the KFCP study and are therefore reported with a high degree of certainty. As per the note to Table 7, however, the emission factors for fires in plantations and croplands are highly uncertain and require further elaboration based on studies of fires in these specific landscape settings. The EF values for second and subsequent fires are assumed to be the same as those for fires on cleared land, i.e. 73 t C ha\(^{-1}\), which is within, but towards the lower end, of the IPCC EF of 103 ± 49 t C ha\(^{-1}\) for prescribed agricultural fires on drained peatlands.

In calculating the fire emission factors we have used the appropriate values for peat bulk density and carbon content derived from the studies described in Section 3. We propose that the same bulk density and C content values should be used when calculating emission factors for all fire events, but we do recognize that for third fires and beyond it would be possible to use a higher bulk density value to take account of the changes that occur in the surface peat as a result of several successive fires. For example, the peat sample analysis undertaken as part of this study suggests that the use of a higher bulk density of 0.132 g cm\(^{-3}\) could be appropriate (peat carbon content remains unchanged at 55%). This finer level of detail could support reporting of fire emissions at Tier 3.
### Table 7: Emission Factors for Peat Fires on Indonesian Peatlands for Reporting at Tier 2 Level

<table>
<thead>
<tr>
<th>Land cover type</th>
<th>EF Fire (t C ha(^{-1}))**</th>
<th>First fire</th>
<th>Second fire</th>
<th>Third fire</th>
<th>Fourth and subsequent fires</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Assumed to occur during first 5 years after drainage, and most likely during the first 2 years.</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>A. Primary peatland forest, never logged and without drainage impacts (i.e. intact and therefore at extremely low risk of fire)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>B &amp; C. Slightly and moderately drained peatland forest (selectively logged with drainage by large canals at ~1000m intervals but not previously burnt)</td>
<td>120</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
<td></td>
</tr>
<tr>
<td>D. Fully degraded peatland (usually burnt at least once with limited regrowth) and drained by large canals at ~1000 m intervals</td>
<td>N/A***</td>
<td>73</td>
<td>27</td>
<td>27</td>
<td></td>
</tr>
<tr>
<td>E. Plantations and croplands, with large canals at intervals of less than 1 km and/or field drains less than 400 m apart [not found in the KFCP study area]</td>
<td>120( \uparrow ) or 73( \downarrow )</td>
<td>73( \infty )</td>
<td>73( \infty )</td>
<td>73( \infty )</td>
<td></td>
</tr>
</tbody>
</table>

** Calculation of EF values assumes a peat bulk density value of 0.121 g cm\(^{-3}\) (average for the upper 0.4 m of peat in forested areas in the KFCP area; value is the average of 0.117 g cm\(^{-3}\) in low-drainage forest and 0.125 g cm\(^{-3}\) in high-drainage forest; see Section 2; a peat carbon content of 55% (average for peat at a depth of 0.2 m in the KFCP area; this value applies to both forested and burnt peatland; and a combustion efficiency of 1.0 (a simplifying assumption based on complete combustion of all organic C).

*** It can be assumed that emissions from the first fire for this land cover type were equivalent to those from land cover B & C.

\( \uparrow \) Value assumes that the conversion is directly from forest; \( \downarrow \) value assumes that the conversion occurs from previously cleared land; \( \infty \) value assumes fire occurs in an established plantation. The plantation fire EF values are all highly uncertain since: a) emissions from land clearing fires will depend on drainage densities and quantity of heavy fuels, and b) emissions from fires in established plantations will depend on land management practices, particularly depth of water table.
7.4 Emission factors and land use change for use in MRV

Figure 15 combines the biological oxidation and fire emission factors in a way that should be easier to apply the values in the context of MRV procedures, accounting for the interrelations between the two emission types and land use change. In preparing this decision matrix, it should be noted that there are some scenarios of land use change that are not explicitly addressed since they were not present within the KFCP study area.

Firstly, there is no class for former forested peatland that is cleared but not burnt. For this scenario, which is probably quite rare in our experience for Indonesian peatlands, we suggest that the biological emissions more than 5 years after drainage are probably closer to Land Cover class C (forest remains) than to D (burnt peatland dominated by ferns/shrubs). This would be a conservative assumption as soil temperatures on deforested peatland will be higher than on forested peatland, which will likely drive a higher rate of emissions from biological oxidation.

Secondly, there is no class for forest regrowth after fire. Again we believe this is best covered by Land Cover class C. This is a conservative assumption since emissions are probably higher than C. This land cover type is likely to be transient in nature, occurring often as a step in development or, if left unmanaged, will be at a high risk of repeat fire.
Figure 15: Pathway for peatland land use change and emission factors

<table>
<thead>
<tr>
<th>Land Cover / Land Use Class:</th>
<th>LULC Class A</th>
<th>Event A to B</th>
<th>LULC Class B</th>
<th>Event B to C</th>
<th>LULC Class C</th>
<th>Event C to D</th>
<th>LULC Class D</th>
<th>Event D to E</th>
<th>LULC Class E</th>
</tr>
</thead>
<tbody>
<tr>
<td>Primary peatland forest</td>
<td></td>
<td>Selective logging / ditch construction</td>
<td>Slightly drained peatland forest</td>
<td>Large canals constructed within 3 km</td>
<td>Moderately drained peatland forest</td>
<td>Fire and deforestation</td>
<td>Fully degraded peatland</td>
<td>Conversion to plantation drainage / fire for land clearance</td>
<td>Plantations and cropland</td>
</tr>
<tr>
<td>Peat GHG emission factors</td>
<td>Biological ox. - Years 1-5¹</td>
<td>0</td>
<td>-</td>
<td>3.95</td>
<td>-</td>
<td>26</td>
<td>-</td>
<td>26</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Biological ox. - &gt; Year 5¹</td>
<td>0</td>
<td>-</td>
<td>3.95</td>
<td>-</td>
<td>7.9</td>
<td>-</td>
<td>4.5</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Fire - 1st fire²</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>120</td>
<td>-</td>
<td>120 or 73</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Fire - 2nd fire²</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>73</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td></td>
<td>Fire - 3rd fire and subsequent²</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>N/A</td>
<td>73</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

¹ For biological oxidation, units are: tC ha⁻¹ yr⁻¹
² For fire, units are: tC ha⁻¹
REFERENCES


