White Paper Number 15 | September 2011

Indirect Effects of Biofuel Production Series

REVIEW OF PEAT SURFACE GREENHOUSE GAS EMISSIONS FROM OIL PALM PLANTATIONS IN SOUTHEAST ASIA

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THE INTERNATIONAL COUNCIL ON CLEAN TRANSPORTATION

ABSTRACT

Palm oil expansion in Southeast Asia is strongly associated with conversion and degradation of peatland. We find that past studies have generally significantly underestimated emissions from palm oil grown on peatland. In particular, this will have resulted in underestimation of the indirect land use change emissions from many biofuels. We suggest that 86 $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} (over 50 years) or 100 $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} (over 25 years) represent the best available estimates of typical emissions from peat decomposition in palm plantations.

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Funding for this work was generously provided by the ClimateWorks Foundation.

EXECUTIVE SUMMARY

Greenhouse gas (GHG) emissions occur at all points in the oil palm (OP) biofuel production chain. This report is concerned with the land use change consequences of expanding palm oil production; specifically, it aims to (i) review the current understanding of likely rates of carbon and GHG emissions from peat decomposition for OP plantations on peat soils in Southeast Asia, and (ii) provide guidance to economic modelers on an appropriate range of values and uncertainties for GHG emissions arising from peat degradation.

Growth in palm oil production, led primarily by Indonesia and Malaysia, has been a key component of meeting growing global demand for vegetable oil over recent decades. This growth has been accompanied by mounting concern over the impact of the OP business on tropical forests and peat swamp forests in particular, where a growing proportion of this production is taking place. Indonesia and Malaysia currently meet more than 85% of global palm oil demand, but in both countries, plantations are increasingly being developed on peat soils.

Tropical peatland is one of the Earth's most spatially efficient carbon sinks and largest long-term repositories of terrestrial organic carbon, with the greatest extent (about 247,778 km^2) and amount of carbon stored (about 68.5 Gt) in Southeast Asia (specifically Indonesia and Malaysia). Tropical peat swamp forest removes carbon dioxide (CO_2) from the atmosphere and stores it in biomass and as deep near-surface peat deposits. The incomplete decomposition of dead tree material under waterlogged, anaerobic conditions has led to the slow but progressive accumulation of thick deposits of peat over millennia, giving this ecosystem a very high carbon density (with typical values of ~150 t C ha^{-1} for forest biomass and ~3000 t C ha^{-1} for the underlying peat).

The tropical peatland carbon and GHG balance is determined largely by the net balance between carbon uptake in photosynthesis and carbon release through ecosystem respiration by vegetation (referred to as autotrophic respiration and resulting in CO_2 emissions from plant foliage and root systems) and by the organisms involved in organic matter decomposition (known as heterotrophic respiration, involving the loss of carbon as CO_2 and CH_4, or methane, by organisms involved in aerobic and anaerobic decomposition of plant litter and peat). A smaller amount of carbon is leached out from the system in drainage runoff. Furthermore, under certain conditions, the cycling of nitrogen (N) makes some tropical peatlands a source of the potent GHG nitrous oxide (N_2O), especially if fertilizer has been added to promote agricultural or plantation productivity.

Development of tropical peatland for agriculture and plantations requires radical changes in the vegetation cover (i.e. replacement of forest by crop

plants) and permanent drainage. Typically, OP cultivation requires drainage depths of 0.6-0.85 m but, in practice, these often exceed 1 m. These changes reduce or, in most cases, remove the carbon sink capacity of the peatland system by (i) lowering of the peat water table, which ensures continuous aerobic decomposition of organic matter (plant litter and peat), resulting in high peat surface CO_2 emissions and (ii) greatly reducing or stopping carbon inputs to the peat from biomass. As a consequence, the peat swamp ecosystem switches from a net carbon sink to a net carbon source (i.e. peat accumulation is replaced by peat degradation), with large carbon losses arising particularly from enhanced aerobic peat decomposition and the loss of any future carbon sequestration by the native peat swamp forest vegetation.

A number of recent publications have addressed the GHG emissions associated with land use conversion of tropical peat swamp forest to OP plantation. All conclude that while carbon losses from biomass replacement and land clearance are considerable, it is the large and sustained CO_2 emissions from drained peat that contribute most to overall emissions and biofuel carbon debts. The values used to estimate peat CO_2 emissions have a wide range (19 to 115 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}) and are derived from a variety of sources, including IPCC defaults and a limited number of scientific studies. Dependency on a limited number of flux studies, combined with inappropriate upscaling, has resulted in systematic underestimation of GHG emissions from OP plantations on tropical peat.

We undertook a rigorous assessment of the empirical foundations, accuracy and validity of emissions estimates, tracing values used back to the original publications and evaluating the approaches and methodologies employed. In general, studies either report (a) direct measurements of gaseous fluxes using closed chambers, or (b) estimates of total carbon loss based on peat subsidence rates (reported as CO_2 equivalents). Each technique has its advantages and disadvantages, largely relating to the spatial and temporal scales of measurement. The majority of studies using closed chamber methods have not been based on sufficient numbers of replicates over sufficient length of time to provide statistically robust flux values or uncertainty ranges. Furthermore, most have not addressed the quantification of CO_2 emissions arising solely from peat decomposition (i.e., excluding emissions arising from root respiration), although some of the data have been used subsequently for this purpose. The second method, subsidence monitoring, is capable of providing a time-integrated measure of the complete carbon balance of a drained peatland. Subsidence is a slow process, however, and thus a key limitation of this approach and of several published studies is that subsidence data need to be collected over a long period (preferably a number of years, although larger numbers of measurements can compensate for shorter periods) and must be accompanied by accurate measurements of peat bulk density and carbon concentration.

CO_2 is the most important GHG emitted from drained peatlands, contributing 98% or more of the total combined global warming potential. Undifferentiated (i.e. peat decomposition plus root respiration) CO_2 emissions calculated through flux measurements vary considerably, ranging from below 30 Mg CO_2 ha^{-1} yr^{-1} to above 100 Mg CO_2 ha^{-1} yr^{-1}. In the few studies that provide differentiated emissions for peat decomposition, values are in the range of 19 to 94 Mg CO_2 ha^{-1} yr^{-1}. Estimates based on subsidence monitoring yield values in the range 45 to 135 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for drainage depths of 0.5 to 1.0 m, and 54 to 115 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for the optimal drainage depth range for OP (60-85 cm). In terms of non-CO_2 GHG emissions, CH_4 fluxes in drained tropical peatland are insignificant relative to losses of CO_2, both in terms of the mass of carbon lost and overall climatic impact. It should be noted, however, that CH_4 emissions from the plantation drainage network may be significant, although this potentially important source of CH_4 remains to be quantified. Similar to CH_4, the likely rates of peat N_2O fluxes in OP plantations remain uncertain, and the limited studies to date are unlikely to have adequately captured the true magnitude and dynamics of emissions, particularly following fertilizer application.

We conclude that a value of **86 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}** (annualized over 50 years) represents the most robust currently available empirical estimate of peat CO_2 emissions from OP and pulpwood plantations, based on combined subsidence measurements and independent closed chamber measurements in the same plantation landscape. Moreover, this estimate explicitly accounts for higher CO_2 emissions observed in the early stages of plantation drainage. For a shorter annualization, the emissions would be higher (see table 1 below).

Table 1: Annualized values for peat carbon losses from plantations over various time scales, accounting for higher rates of emissions in the years immediately following drainage (values derived from Hooijer et al., 2011).

NUMBER OF YEARS	CARBON LOSS (Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1})
5	178
10	121
20	106
25	100
30	95
40	90
50	86

In terms of an uncertainty range, we suggest that likely peat CO_2 emissions should be represented by the minimum and maximum values of **54** to **115 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}** for the typical OP drainage depth range of 0.6 to 0.85 m. It should be noted that none of these values explicitly consider local factors promoting GHG emission other than water depth (e.g., fertilization, land use history) or regional geographical variations. The adoption of the best estimate and full uncertainty range suggested here will, however, lead to reduced uncertainty in future assessments conducted at the regional scale.

The majority of previous studies aiming to assess GHG emissions from OP production systems on tropical peatlands have at best based their analyses on values below or towards the lower end of this range, and in all likelihood have significantly underestimated CO_2 emissions from drained peats. In terms of biofuel production, it is likely that the true magnitude of the biofuel carbon debt for OP feedstocks produced on tropical peatlands is more substantial than has been previously assumed.

ACKNOWLEDGMENTS

This review was conducted by Mr. Ross Morrison and Dr. Susan E. Page of the University of Leicester, United Kingdom, with additional contributions from

- Dr. Aljosja Hooijer (Deltares, the Netherlands)
- Professor Jack Rieley (University of Nottingham, United Kingdom)
- Dr. Jyrki Jauhiainen (University of Helsinki, Finland)

The report was commissioned by Dr. Chris Malins of the International Council on Clean Transportation (ICCT). The research was funded by the ICCT, Washington, DC, USA.

The authors recognize contributions from the following individuals and institutions:
Dr. Stephanie Searle (ICCT)
Tina Godfrey (University of Leicester)

This report should be cited as:

Page, S. E., Morrison, R., Malins, C., Hooijer, A., Rieley, J. O. & Jauhiainen, J. (2011). *Review of peat surface greenhouse gas emissions from oil palm plantations in Southeast Asia* (ICCT White Paper 15). Washington: International Council on Clean Transportation.

This review has been produced as a consultancy report; a scientific version of the research presented here will also be submitted for publication in the peer-reviewed academic literature in due course.

GLOSSARY

Aerenchyma	air-conducting tissue present in some plants
Aerobic	with oxygen
Anaerobic	without oxygen
Anoxic	process or environment that is without oxygen
Autotrophic	organisms capable of synthesizing their own food
Benthic	at the bottom of a body of water
Cramer Commission	headed by Dutch Environment Minister Jacqueline Cramer, the commission reported to the Dutch government on biofuel sustainability in 2007
Denitrification	conversion of nitrate to nitrous oxide or nitrogen gas
Ebullition	bubbling
Ecosystem respiration	the combined loss of CO_2 to the atmosphere from all the various respiring elements of the ecosystem (i.e., the sum of autotrophic and heterotrophic respiration)
FLUXNET	global network of micrometeorological eddy covariance towers
Heterotrophic	organism that is not capable of providing its own food source
Lignin	a complex organic polymer derived from wood
Methanogens	microbes that form methane
Microtopography	small-scale, local elevational features of the landscape; on a tropical peatland, hummocks and hollows have an elevation range of ~50-75 cm
Minerotrophic	peatland that receives water through surface runoff and/or groundwater
Mycorrhizae	symbiotic fungi living in association with plant roots
Nitrification	the conversion of ammonium to nitrate
Ombrotrophic	rain-fed peatland
Oxic	process or environment that uses oxygen
Peat respiration	the total amount of CO_2 emitted at the peat surface; a combination of root and root-associated mycorrhizal (autotrophic) respiration and the respiration of heterotrophic macro- and microorganisms involved in the decomposition of organic matter (plant litter and peat)
Pneumatophore	specialized tree root structure that grows out from the water surface and facilitates the aeration necessary for root respiration
Rhizosphere	the root zone
Root respiration	the amount of CO_2 released from the rhizosphere as a result of the metabolism of roots and root-associated mycorrhizal fungi
Trenching	severing roots prior to chamber measurements of CO_2

ABBREVIATIONS

asl	Above sea level
^{14}C cal ka BP	Thousands of years before present (based on calibrated carbon-14 isotope decay rate)
CH_4	Methane
CO_2	Carbon dioxide
DOC	Dissolved organic carbon
EC	Eddy covariance
GHG	Greenhouse gas
ICCT	The International Council on Clean Transportation
IFPRI (MIRAGE)	International Food Policy Research Institute (Modelling International Relationships in Applied General Equilibrium)
iLUC	Indirect land use change
IPCC	Intergovernmental Panel on Climate Change
N_2	Nitrogen (gas)
N_2O	Nitrous oxide
NH_4^+	Ammonium
NECB	Net ecosystem carbon balance
NEE	Net ecosystem exchange
NO_3^-	Nitrate
NO_x	Generic term for nitrogen oxides
NPP	Net primary production
OP	Oil palm
POC	Particulate organic carbon
TOC	Total organic carbon
USDA	U.S. Department of Agriculture

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1 INTRODUCTION

Greenhouse gas (GHG) emissions occur at all points in the oil palm (OP) biofuel production chain; however, this report is only concerned with the land use change component, and specifically that related to greenhouse gas emissions from the long-term peat carbon store.

Palm oil is the world's principal source of vegetable oils and fats, exceeding soybeans. Growth in palm oil production, led primarily by Indonesia and Malaysia, has been a key component of meeting growing global vegetable oil demand over recent decades (Carter, Finley, Fry, Jackson, & Willis, 2007; Fargione, Hill, Tilman, Polasky, & Hawthorne, 2008). With an increasing number of countries adopting mandates and incentives for biofuels, palm oil has found a new market as a biodiesel feedstock (Zhou & Thomson, 2009). This growth has been accompanied by mounting concern over the impact of the OP industry on tropical forests and carbon-dense peat swamp forests in particular. These concerns go beyond the direct risk that palm oil specifically destined for biodiesel will be grown at the expense of peat swamp forest to include the likelihood that palm oil production will expand to fill the resulting supply shortage as other vegetable oils (soy, rapeseed, sunflower) are increasingly committed to biodiesel production, thus leading to indirect land use change (iLUC). Attempting to quantify the carbon consequences of this iLUC, several government regulators (notably the U.S. Environmental Protection Agency[1], the European Commission,[2] and the California Air Resources Board[3]) along with other institutions have used economic modeling to predict where land use change is likely to take place (e.g. Lapola et al., 2010; Searchinger et al., 2008).

Traditionally, expansion of palm oil production in Southeast Asia has been associated with forest loss (Koh & Wilcove, 2007, 2008), and therefore the amount of increased vegetable oil demand that will be met by palm oil has been seen as an important parameter in the carbon assessments by these models. Despite the high profile of deforestation in Southeast Asia, however, these models have generally not included emissions from drained peatlands, despite the fact that an increasing proportion of OP plantations are located on peatland and that emissions from drainage-impacted peat can be significantly greater than emissions from above ground biomass clearance. In 2010, the European Commission Joint Research Center (JRC) drew attention to this gap in the existing modeling framework and the likelihood that correctly including emissions from peat soils could substantially affect the carbon balance of biodiesel. Even in the one model that had included peat explicitly (IFPRI MIRAGE, see Al-Riffai, Dimaranan & Laborde, 2010), an emissions value of 22.5 t CO_2-eq ha^{-1} yr^{-1} was used, which JRC (2010)

1 http://www.epa.gov/otaq/fuels/renewablefuels/index.htm

2 http://ec.europa.eu/energy/renewables/consultations/2010_10_31_iluc_and_biofuels_en.htm

3 http://www.arb.ca.gov/fuels/lcfs/lcfs.htm

notes is a considerable underestimate. This report aims to clarify the issues surrounding GHG emissions from drained peat, and provide guidance to economic modelers and others on an appropriate range of values to use and uncertainties to account for when including peat emissions figures in calculations.

Although not addressed in this report, in addition to increasing land-atmosphere carbon emissions, growth in demand for palm oil has also been cited as a driver of the loss and fragmentation of both primary and secondary (logged) forests and reduced provision of forest ecosystem services, including biodiversity, watershed protection, and flooding and erosion control (Butler & Laurance, 2009; Danielsen et al., 2009; Fitzherbert et al., 2008; Koh & Wilcove, 2008). Furthermore, the expansion of OP has been accompanied by both positive and negative social and economic outcomes. While plantation estates and associated downstream processing and service facilities provide employment opportunities (Ministry of Forestry, 2001) and contribute to local and national economic growth and poverty reduction (Susila, 2004), they have also been responsible for both land and labor disputes, particularly where forests subject to indigenous land rights have been expropriated (Phalan, 2009).

1.1 Aims of the report

The primary aims of this report are to:

a) review the current understanding of likely rates of carbon and greenhouse gas emission from OP plantations on peat soils, and of the best-case scenario for reduced peat GHG emissions with best agricultural practice; and
b) identify best typical estimates and uncertainty profiles for the rate of peat GHG[4] emissions from palm oil plantations on peat soils in Southeast Asia.

1.2 Approach

This report presents a comprehensive review of the available literature relating to GHG emissions from OP plantations on peatlands in Southeast Asia. In order to assess fully the implications of land use conversion of peat swamp forest to OP plantation, it is necessary to describe the carbon cycle processes and associated GHG emission dynamics operating in both natural peat swamp forest and plantation systems. Furthermore, to evaluate the accuracy of current estimates, it is essential to assess and compare the empirical methods currently used in quantifying GHG emissions from tropical peatlands. Where possible, the review draws on findings from peer-reviewed scientific literature and official reports; however, publications and other sources from the gray (non-peer reviewed) literature are cited where

4 The greenhouse gases addressed in this report are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O).

necessary. Additionally, while the focus is on OP plantations on tropical peatlands, information relating to the peatlands of the temperate and boreal climatic zones is referred to where appropriate. The review is structured to address the following topics:

a) a brief overview of the status of peatlands in Southeast Asia, detailing the best currently available estimates of the areal extent of and carbon storage in peatlands in the region and the current extent of OP plantation development on tropical peatland.

b) the functioning of natural peat swamp forest and tropical peatlands that have been converted to OP plantations, specifically relating to ecosystem carbon cycle processes and associated carbon GHG dynamics, but also assessing the release of N_2O.

c) an overview of the methods currently used in estimating peat surface carbon losses and GHG emissions from tropical peatlands, and from tropical peatlands that have undergone land use change for OP plantation development.

d) a critical review of the literature relating to carbon losses and GHG emissions from OP plantations on peat in Southeast Asia, focusing on both the empirical data and their use in estimating emissions.

2 BACKGROUND

2.1 Peatlands in Southeast Asia

This report relates to lowland tropical peatlands in Southeast Asia; specifically, to those peatlands that have experienced land use change for OP plantation development. Tropical peatland is one of the Earth's most spatially efficient carbon sinks and largest long-term repositories of terrestrial organic carbon (Page, Rieley & Banks, 2011). Although tropical peatlands are found in all humid tropical regions, the largest area and carbon storage is located in Southeast Asia (Page et al., 2011). The best currently available estimates indicate that peatlands in this region cover an area of about 247,778 km^2 and store approximately 68.5 Gt carbon in the peat (Page, et al., 2011). This regional peat carbon store is estimated to be 77% of the carbon in all tropical peatlands and about 11-14% of the global peatland carbon pool (Page et al., 2011).

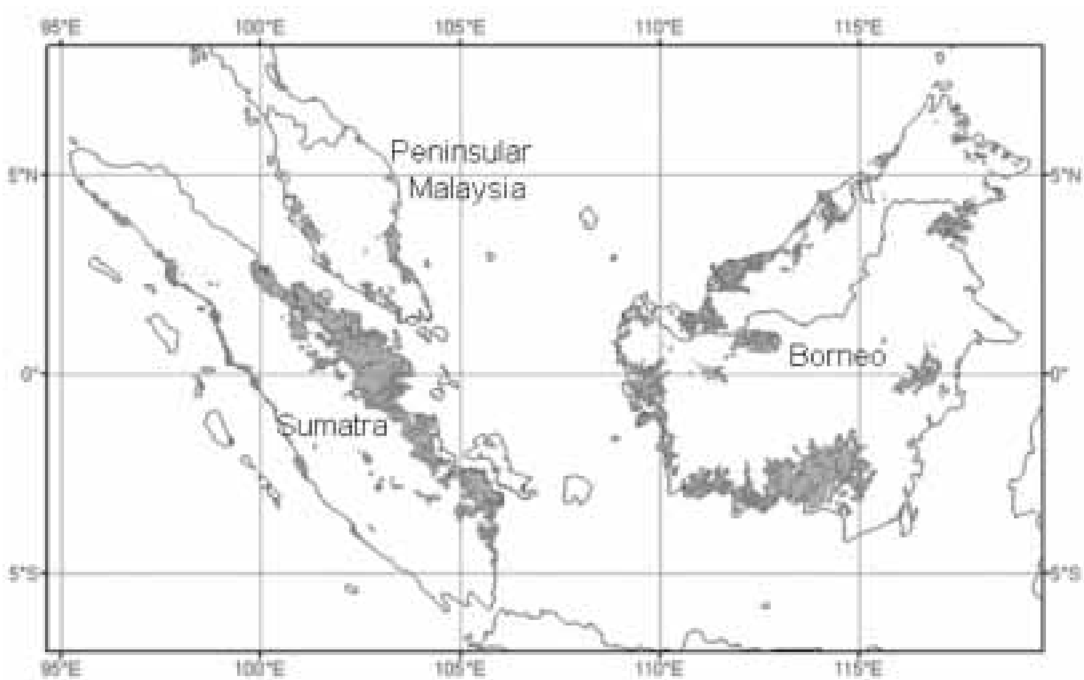

Figure 1: The distribution of the main peatland deposits of Southeast Asia. Most peatlands occur on the islands of Sumatra and Borneo (Kalimantan, Sarawak, and Brunei) and in peninsular Malaysia (derived from the International Council on Clean Transportation, in press)

In SE Asia (Fig. 1), extensive peatland carbon resources are located in the countries of Indonesia (57.4 Gt C), and Malaysia (9.1 Gt C), with less extensive areas in Papua New Guinea, Thailand, Cambodia and the Philippines (Delft Hydraulics, 2006, 2010; Page et al., 2011). The majority of these peatlands are located in coastal and subcoastal lowlands less than 20 m above sea level. Some areas of peatland are found at higher elevations, but these represent only a small fraction of the regional peatland area and carbon pool (Delft Hydraulics, 2006, 2010; Page et al., 2011) and are unlikely to be converted to OP plantations owing to their inaccessibility and climate. They are not considered in this report.

Most SE Asian lowland peatlands are ombrotrophic (rain-fed), although a few are minerotrophic (receiving surface runoff and/or groundwater). A combination of low topographic relief, impermeable substrates, and high effective rainfall have provided conditions suitable for slow decomposition of organic material and the accumulation of thick (often >10 m) dome-shaped deposits of woody peat (Delft Hydraulics, 2006; Jaenicke, Rieley, Mott, Kimman, & Siegert, 2008; Page et al., 2009). Peatlands are widespread along the region's maritime fringes, in deltaic areas, and further inland at slightly higher elevations (5-15 m asl), where they occur along river valleys and in low-altitude, watershed positions. In addition, some isolated basin deposits have formed in and around lakes.

2.2 Peat swamp forest

Like all tropical forests, peat swamp forest stores large amounts of carbon in plant biomass, with typical values in the range of 100-250 t C ha^{-1} (Murdiyarso, Hergoualc'h & Verchot, 2010; Page et al., 2011). The incomplete decomposition of dead tree material, especially roots, has led to the slow but progressive accumulation of partially decomposed organic material (peat) over millennia and has given this ecosystem a very high carbon density. The best estimate for Indonesian peatland of 2772 t C ha^{-1} is based on an average peat thickness of 5.50 m (Page et al., 2011). Peat (and carbon) accumulates as a result of a positive net imbalance between high tropical ecosystem primary production and incomplete organic matter decomposition in permanently saturated soil conditions (Hooijer et al., 2010; Wösten, Clymans, Page, Rieley, & Limin, 2008).

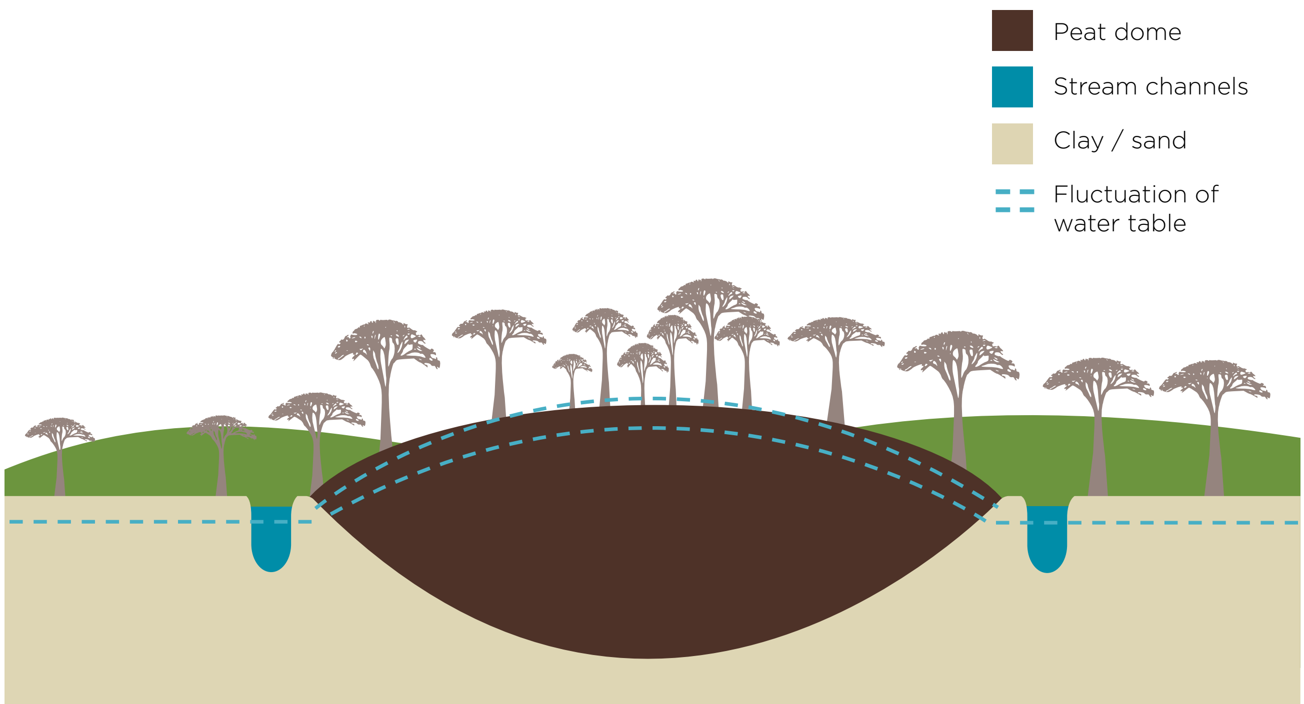

Figure 2: Schematic diagram of a tropical peat dome; the water table will be at or above the peat surface for much of the year, with limited drawdown during the dry season (typical range 30-40 cm).

In a natural state (Fig. 2; Fig. 3), peat swamp forests are characterised by dense forest vegetation, thick (up to 20 m) peat deposits, and a groundwater table that is at or close to the peat surface throughout the year (Hirano, Jauhiainen, Inoue, & Takahashi, 2009; Page et al., 2004; Takahashi et al., 2002; Wösten et al., 2008). Surface peat is aerobic (unsaturated) only during relatively dry periods when the water table falls below the surface (Hirano et al., 2007, 2009; Jauhiainen et al., 2005; Jauhiainen, Vasander, Rieley, & Page, 2010). Tropical peat has low bulk density (approximately 0.1 g cm^{-3}) compared to mineral soils, being formed of approximately 10% tree remains and 90% water (Hooijer et al., 2010), and is 50-60% carbon by dry weight (Neuzil, 1997; Page et al., 2011; Page, Rieley, Shotyk, & Weiss, 1999). The proportion of lignin in tropical peat may be up to 75% on a dry mass basis (Hardon & Polak, 1941, as cited in Andriesse, 1988). The accumulation of peat deposits over time has isolated the peatland surface from mineral-rich groundwater; hence, ombrotrophic tropical peatlands are generally acidic and nutrient-poor, receiving all water and nutrients from precipitation (Jauhiainen et al., 2010; Page et al., 1999, 2004; Wösten et al., 2008).

Figure 3: View over the peat swamp forest canopy, Kalimantan, Indonesia (left); interior of peat swamp forest, Sumatra, Indonesia (right) (Images: S. Page).

The long-term maintenance of peat (and carbon) depends on both continuous inputs of organic matter and incomplete decomposition, a consequence of anaerobic conditions within the peat deposit. For peat accumulation to occur, the average rate of carbon sequestration must exceed that in decomposition losses. (Jauhiainen, Limin, Silvennoinen, & Vasander, 2008; Jauhiainen et al., 2005, 2010).

2.3 Role of tropical peatlands in the global carbon cycle

Only a few peatlands in Southeast Asia have been investigated for peat structure, age, development, and rates of peat and carbon accumulation (e.g., Brady, 1997; Neuzil, 1997; Page et al., 2004; Wüst & Bustin, 2004). The onset and development of peat deposits in this region range from the Late Pleistocene through to the Holocene (~40 ^{14}C cal ka BP to 3.5-6 cal ka BP; Page et al., 2004). Radiocarbon dating of peat material reveals a long-term median peat accumulation rate of ~1.3 mm yr^{-1} (i.e., 67 g C m^2 yr^{-1}, assuming a peat bulk density of 0.09 and 56% C content), which is about two to 10 times the rate for boreal and subarctic peatlands (0.2-0.8 mm yr^{-1}; Gorham, 1991; Page, Wüst & Banks, 2010; Page et al., 2004).

The existence of extensive peat deposits in Southeast Asia demonstrates that these ecosystems have functioned as a large net sink for atmospheric carbon at millennial time scales (Page et al., 2010, 2004). Currently, however, the vast majority of the peatlands in Southeast Asia are to some extent degraded as a result of anthropogenic land use change and are emitting accumulated carbon faster than it is being sequestered (Hooijer et al., 2010, 2011; Jauhiainen et al., 2010; Page et al., 2011). Deforestation, drainage, large-scale conversion to plantation agriculture, and regular fires have all resulted in increased carbon transfer to the environment and loss of carbon sequestration function. Optimum carbon storage in tropical peat requires a combination of high vegetation biomass (carbon sequestration potential), a water table that is near to or above the peat surface for most of the year, and a much-reduced rate of organic matter decomposition. Drainage and other disturbances lead to increased surface peat aeration and decomposition and carbon losses. Only two studies have been carried out so far on the annual ecosystem peat swamp forest carbon balance. Suzuki, Nagano & Waijaroen, (1999) found a ~530 g C m^{-2} yr^{-1} (5.3 t C ha^{-1} yr^{-1}) net negative carbon balance (indicating peat accumulation, assuming that the biomass in mature forest did not vary over the study period) in primary peat swamp forest in To-Daeng, Thailand, in a typical wet year. In comparison, Hirano et al. (2007) found a ~600 g C m^{-2} yr^{-1} (~6 t C ha^{-1} yr^{-1}) net positive carbon balance (peat loss) in drained peat swamp forest in Central Kalimantan, Indonesia, during the dry El Niño year of 2002, although this loss was nearly halved in wet years due to a higher water table.

Current carbon emissions from drained and fire-affected peatlands in Southeast Asia have been estimated to be of the order of ~360 Mt C yr^{-1}: ~170 Mt C yr^{-1} from drainage-related peat decomposition (Delft Hydraulics, 2006) and 190 Mt C yr^{-1} from peat fires (Page et al., 2002; van der Werf et al., 2008). Losses on this scale contribute significantly to atmospheric carbon loading and anthropogenic climate change processes (Page et al., 2011, 2002); the long-term instability of the large amount of carbon stored in tropical peatlands is of major concern within the context of contemporary climate change (Raupach & Canadell, 2010).

2.4 Oil palm plantations in Southeast Asia

OP plantation establishment and palm oil production has grown rapidly in Southeast Asia, with Indonesia and Malaysia currently meeting more than 85% of global palm oil demand (Danielsen et al., 2009; Fargione et al., 2008). In 2006-07, production of palm oil in these two countries was 31.9 x 10^3 metric tonnes, rising to 41.1 x 10^3 metric tonnes in 2010-11 (Foreign Agricultural Service, 2011). This increase has contributed to deforestation across the Southeast Asian region and is increasingly focused on peat soils (Fig. 4). Between 1990 and 2007, 5.1 Mha of the total 15.5 Mha of peatland in peninsular Malaysia and the islands of Borneo and Sumatra was deforested, drained, and burned, while most of the remainder was logged intensively (Langner & Siegert, 2009; Miettinen & Liew, 2010). Over the same period, industrial OP and pulpwood (*Acacia*) plantations expanded dramatically, from 0.3 Mha to 2.3 Mha (likely comprising 2.1 Mha of OP and 0.2 Mha of *Acacia*), an increase from two percent to 15% of the total peatland area (Miettinen & Liew, 2010).[5] The estimated area of 2.1 Mha for industrial-scale OP plantations on peat in peninsular Malaysia, Sumatra, and Borneo in 2010

5 The recent estimate of ~0.9 Mha of OP plantation on peatland by Koh, Miettinen, Liew, & Ghazoul (2011) is for mature OP that was planted before 2002, with young OP plantations not captured by their analysis method (Paoli et al., 2011).

was recently confirmed by a mapping exercise using high-resolution remote sensing imagery (ICCT, in press).

The area of OP on peat in Malaysia is estimated to have increased from 0.38 Mha in 2000 to 0.53 Mha in 2010 (ICCT, in press), with most of this increase occurring in Sarawak. Over the period 2000-2010, ca. 0.42 Mha of peatland was opened in Sarawak for OP production and expansion, with a total area of 1.2 Mha anticipated to be opened by the year 2020. In Indonesia, OP plantations on peat are estimated to cover 1.3 Mha (ICCT, in press), with around 1.0 Mha occurring in Sumatra and 0.3 Mha in Kalimantan. Projections indicate an increase to a total area of 2.5 Mha in Sumatra and Kalimantan by the year 2020 (ICCT, in press).

Figure 4: Oil palm plantations on peat: (left) immature stage (Image: S. Page) and (right) mature stage, note the leaning trunks owing to low load-bearing capacity of peat soils (Image: J. Jauhiainen).

2.5 Carbon and greenhouse gas balance of tropical peatlands

The tropical peatland carbon and GHG balance is determined largely by the net balance between carbon uptake in photosynthesis and carbon release through ecosystem respiration by: (a) vegetation (referred to as autotrophic respiration and resulting in carbon dioxide [CO_2] emissions from both plant foliage and root systems) and (b) by the organisms involved in organic matter biological decomposition (known as heterotrophic respiration, involving the loss of carbon as CO_2 and CH_4, or methane, by organisms involved in aerobic and anaerobic decomposition of organic matter, comprising plant litter, roots and their exudates, dead animals, fungi, bacteria and the peat itself; Fig. 5). An additional, smaller amount of carbon is leached out from the system in drainage runoff as dissolved organic carbon (DOC) or particulate organic carbon (POC; Moore, Gauci, Evans, & Page, 2011). Furthermore, under certain conditions, the cycling of nitrogen (N) makes some tropical peatlands a source of the potent greenhouse gas nitrous oxide (N_2O), especially if fertilizer has been added to promote agricultural or plantation productivity (Germer & Sauerborn, 2008; Jauhiainen et al., 2011; Melling, Goh, Beauvais, & Hatano, 2007a; Murdiyarso et al., 2010).

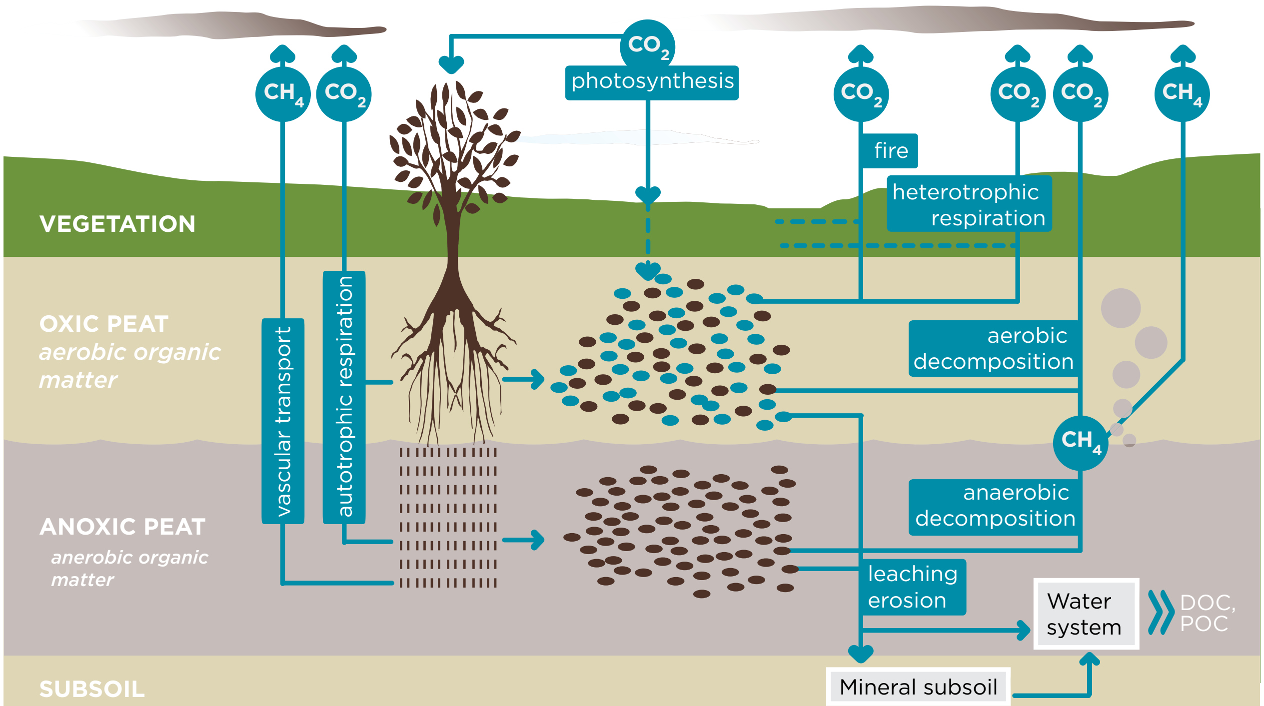

Figure 5: Schematic representation of carbon cycle processes, flow paths and stores in tropical peat (after Jauhiainen, Heikkinen, Martikainen, & Vasander, 2001).

The following sections provide an overview of the processes and controls driving carbon and GHG fluxes in tropical peat swamp forests. It is important to note (a) that carbon cycle and GHG processes are highly dynamic and vary at all spatial and temporal scales owing to regional and local variations in macro- and micro-climate and hydrology, as well as localised variations in vegetation and peat decomposition dynamics (Hooijer et al., 2011; Jauhiainen, Hooijer, & Page, in review; Jauhiainen et al., 2005, 2010; JRC, 2010; Renewable Fuels Agency, 2010); and (b) that in terms of emissions and global warming potential, CO_2 is the most important gas emitted from drained peatlands, contributing 98% or more of the total combined global warming potential (GWP) of CO_2, CH_4 and N_2O (Couwenberg, Dommain & Joosten, 2010; Jauhiainen et al., 2011). This report focuses on carbon losses and GHG emissions arising from biological processes; it excludes fires, which are an additional source of CO_2, CO, CH_4 and NO_x formed during combustion of biomass and peat.

2.5.1 CARBON DIOXIDE (CO_2)

In peat swamp forest, atmospheric CO_2 is fixed (reduced to carbohydrate) during photosynthesis by the vegetation. Some of this gross primary production (GPP) is returned quickly to the atmosphere via the autotrophic respiration of the vegetation itself and mycorrhizal fungi associated with tree roots, while the remaining net primary production (NPP) is manifest as increments to the above- and below-ground forest biomass. Tree litter, in the form of leaves, branches and occasional boles, is deposited onto the peatland surface, whilst the turnover of fine roots results in deposition of organic matter directly within the peat profile. If oxic conditions (low water table) are present in the upper peat profile, decomposition of this organic matter (carbon) and some of the surface peat proceeds along aerobic pathways by heterotrophic bacteria and fungi, which release CO_2 to the atmosphere. If the water table is at or above the peat surface, then aerobic peat decomposition is in practice prevented and decomposition of deposited organic matter takes place at a reduced rate along anaerobic pathways by methanogenic heterotrophic bacteria, resulting in the release of CH_4 (see below). The undecomposed fraction is incorporated into the surface peat layer. As tropical peatlands are typically ombrogenous (entirely rain-fed), fluctuations in the peatland water table are largely a function of seasonal changes in rainfall and the capacity of the peatland surface to slow radial water flow, e.g. through the presence of a hummock-hollow surface microtopography (Jauhiainen et al., 2010; Page, Hooijer, Rieley, Banks, & Hoscilo, in press). The accumulation of peat and long-term storage of carbon occurs when a fraction of the organic matter produced by the forest vegetation enters the saturated peat zone below the permanent water table.

To date, there have been few studies addressing whole ecosystem CO_2 budgets for peat swamp forests in Southeast Asia (Hirano et al., 2007; Suzuki et al., 1999) with only one addressing CO_2 emissions from an intact primary forest in Thailand (Suzuki et al., 1999). Most peer-reviewed scientific literature presents gaseous CO_2 emissions from the peat surface only (Ali, Taylor & Inubushi, 2006; Furukawa, Inubushi, Ali, Itang, & Tsuruta, 2005; Hadi et al., 2005; Hirano et al., 2009; Inubushi, Furakawa, Hadi, Purnomo, & Tsuruta, 2003; Jauhiainen et al., in review, 2008, 2005; Melling, Hatano & Goh, 2005a).

The key processes influencing CO_2 exchange between tropical peat swamp forest and the atmosphere are the rates of NPP (photosynthesis minus autotrophic respiration) and CO_2 emissions from the peat soil (Hirano et al., 2009, 2007; Suzuki et al., 1999). The rate of NPP depends largely on the structure (density, girth, height) of the forest vegetation, which varies across the peat dome in response to hydrology, peat thickness, and nutrient status (Page et al*.,* 1999; :Fig. 2), together with any natural or anthropogenic disturbances (e.g., fire or logging). Heterotrophic CO_2 emissions from the peat are influenced by microbial population dynamics, the quality (decomposability) and quantity of the organic material available for decomposition (litter, peat and root exudates), which in turn is a function of past and present vegetation dynamics, microbial responses to hydrology (water table depth and soil moisture), air and peat temperature, and peat nutrient status (Brady, 1997; Hirano et al., 2009; Jauhiainen et al. in review, 2008, 2005; Yule & Gomez, 2009). In general, increased availability of labile (i.e., readily decomposable) organic material, the lowering of the peatland water table (increased aeration), increased temperatures, or higher peat nutrient status (e.g., as a result of fertilization) will all enhance peat decomposition rates and CO_2 emission from the peat.

Development of tropical peatland for agriculture and plantations requires radical changes in the vegetation cover (i.e., replacement of forest by crop plants) and permanent drainage. These changes reduce or, in most cases, remove the carbon sink capacity of the peatland system by (a) lowering of the peat water table, which ensures continuous aerobic decomposition of organic matter, increased peat temperature, and hence enhanced aerobic heterotrophic respiration and higher peat surface CO_2 emissions for much if not all of the year (Delft Hydraulics, 2009; Hooijer, Haasnoot, van der Vat & Vernimmen, 2008; Jauhiainen, in review) and (b) greatly reducing or stopping carbon inputs to the peat from biomass. As a consequence, the peat swamp ecosystem switches from a net carbon sink to a net carbon source (i.e., peat accumulation is replaced by peat degradation), with large CO_2 losses from enhanced aerobic peat decomposition (Delft Hydraulics, 2006; Hooijer et al., 2010, 2011; Jauhiainen et al., in review, 2008).

2.5.2 METHANE (CH_4)

Methane (CH_4) is produced during the decomposition of organic matter by methanogenic bacteria under anaerobic conditions (Jauhiainen et al., 2005). CH_4 diffusing towards the atmosphere may also be oxidized to CO_2 by methanotrophic bacteria at times when oxic conditions are present in the upper peat profile (e.g., during the dry season when the water table falls below the surface; Couwenberg et al., 2010; Inubushi et al., 2003; Jauhiainen et al., 2008, 2005). Key factors controlling CH_4 production are water table, temperature, and the quality (decomposability) and quantity of litter and root exudates, which provide an additional carbon source (Hirano et al., 2009; Inubushi et al., 2003; Melling et al., 2005a). Peat CH_4 emissions typically increase when the water table rises, as fresh litter becomes available to methanogens and methanotrophy is reduced. Conversely, CH_4 emissions decrease when the water table falls, owing to lower CH_4 production, increased methanotrophy and conversion of CH_4 liberated from waterlogged peat deeper in the profile to CO_2 as it passes through the unsaturated, oxic surface peat (Inubushi et al., 2003; Jauhiainen et al., 2008, 2005; Melling et al., 2005a; Couwenberg et al., 2010). CH_4 fluxes in tropical peat are low compared to those in northern peatlands, a fact that has been attributed to the much higher amount of lignin-derived carbon compounds in the former (Couwenberg et al., 2010).

2.5.3 NITROUS OXIDE (N_2O)

The cycling of nitrogen in tropical peatlands can result in the production of the potent greenhouse gas nitrous oxide (N_2O). Emissions of N_2O may be a significant component of the peat greenhouse gas balance, as N_2O has a global warming potential 298 times that of CO_2 on a 100-year time scale (IPCC, 2006). N_2O is a byproduct of the biological process of nitrification (the conversion of ammonium, NH_4^+, to nitrate, NO_3^-), and as an intermediate of denitrification (conversion of NO_3^- to N_2O or N_2) (Jauhiainen et al., 2011). Factors that influence N_2O production include peat temperature, soil moisture and water-filled pore space, and the nitrogen status of the peat (Hadi et al., 2000; Inubushi et al., 2003; Melling et al., 2007a). N_2O emissions are typically erratic, often occurring as pulse events, spatially variable and hard to predict, as they are generally not well correlated with any single environmental factor. Providing other conditions allow, maximum N_2O emissions from peat usually occur under conditions intermediate between aerobic and anaerobic; changes in peat hydrology may therefore have a major influence on N_2O emissions (Jauhiainen et al., 2011). As drainage increases peat mineralization rates and NO_3^- availability, the potential for N_2O production becomes significantly greater (Hadi et al., 2000; Jauhiainen et al., 2011). Similarly, applications of nitrogen fertilizer generally serve to increase rates of N_2O emission (Hadi et al., 2000; Takakai et al., 2006), although this increase can be a transient phenomenon if high nitrogen availability is not maintained (Jauhiainen et al., 2011).

2.5.4 DISSOLVED AND PARTICULATE ORGANIC CARBON (DOC AND POC)

In addition to gaseous emissions from peat, carbon is also lost from peatlands via movement of dissolved and particulate organic carbon (DOC and POC, respectively) along hydrological pathways (Baum, Rixen & Samiaji, 2007; Moore et al., 2011). The total loss of fluvial carbon is termed total organic carbon (TOC). In general terms, the distinction between DOC and POC is based on the size fraction of the organic material: DOC is defined as material that passes through a 0.45 μm filter, whilst POC is the larger particulate material retained (Moore et al., 2011). Fluvial carbon outflow is increasingly recognized as an important component of tropical peatland carbon budgets (Moore et al., 2011). As these authors note, however, the ultimate fate of carbon exported from peatlands via fluvial pathways remains largely unquantified and it is not known how much of this carbon is converted and emitted as CO_2 and/or CH_4 during fluvial transport, nor how much remains climatically inert in long-term storage in riverine, estuarine, and benthic sediments (Battin, Luyssaert, Kaplan, Richter, & Tranvik, 2009; Moore et al., 2011).

3 IMPACTS OF OIL PLANTATION DEVELOPMENT

In their natural state, the wet and nutrient poor conditions of tropical peatlands are unfavorable for OP plantations or, indeed, most forms of agricultural production. To be productive and competitive with more favorable mineral soils, the development of OP plantations on tropical peats requires considerable anthropogenic modification of these ecosystems, including removal of forest trees; clearance of remaining debris, often accomplished using fire (JRC, 2010; Murdiyarso et al., 2010); and drainage to create appropriate soil moisture conditions for the OP crop (Delft Hydraulics, 2006; Hooijer et al. 2010). Drainage is accomplished by construction of a network of deep canals and shallower ditches to facilitate rapid removal of water from the OP rooting zone (Alterra, 2008; Delft Hydraulics, 2006; Hooijer et al. 2010). Additionally, the peat surface is often compacted by running heavy vehicles over it to improve the load-bearing capacity of the substrate and increase the stability of the palms (Alterra, 2008; Melling, Hatano, & Goh, 2007b). Application of nitrogen fertilizer in the form of urea is necessary to optimize the productivity of the palms growing on the drained peat (Melling et al., 2007a; Murdiyarso et al., 2010). OP plantations typically operate on a 25-year production cycle, because harvesting costs become increasingly restrictive as tree height increases and palm kernel productivity declines (Corley & Tinker, 2003). On completion of a productive cycle, the plantation is renewed by land clearance, renewed drainage, and replanting.

3.1 Impacts of oil palm plantation development on peatland carbon and greenhouse gas emissions

The conversion of peat swamp forest to OP plantation has major impacts on the carbon and GHG balance of tropical peatland (Hooijer et al, 2010, 2011; Murdiyarso et al., 2010), because significant amounts of stored carbon are lost at all stages of the land use conversion and plantation management processes (Alterra, 2008; Murdiyarso et al., 2010). In the case of natural forest conversion, there is a one-off loss of carbon from the original forest biomass, although biomass carbon losses may be lower when other land cover types, such as degraded (logged-over) forest or, less commonly, existing agricultural lands are converted (Paoli et al., 2011). The replacement oil palms store much less carbon in their biomass than either intact or degraded peat swamp forest; Murdiyarso et al. (2010), for example, suggest values of 254.5 and 24.2 Mg C ha^{-1} for natural peat swamp forest and OP, respectively. Although land clearance by fire has been banned for some years, it is still in widespread use, particularly by smallholders lacking access to heavy plant machinery (JRC, 2010; Suyanto, Applegate, Permana, Khususiyah, & Kurniawan, 2004) but also by some larger plantation operators, especially in Indonesia (Hergoualc'h & Verchot, 2011). Carbon losses attributable to land clearance by fire are very large, particularly when surface peat ignites (Delft Hydraulics, 2006; Hooijer et al., 2010; Murdiyarso et al., 2010; Page et al., 2002), and represent a carbon stock loss additional to that arising from forest biomass combustion (Germer & Sauerborn, 2008; Gibbs et al., 2008; Murdiyarso et al., 2010) and peat carbon losses in decomposition following drainage. In cases where burning is not used in land clearance, forest debris is typically piled up on the land surface and left to decompose and oxidize to CO_2 over time (JRC, 2010). The most significant and long-term effects on the carbon and GHG balances of plantations, however, are caused by drainage-related changes to peat carbon stocks, together with the loss of carbon sequestration by the native peat swamp ecosystem (Fargione et al., 2008; Germer & Sauerborn, 2008; Hooijer *et al*., 2010, 2011; Murdiyarso et al., 2010; Page et al., 2011).

3.2 Impacts of peatland drainage

The accumulation and long-term maintenance of peat carbon requires a continuous supply of organic matter and a water table that is at or close to the peatland surface throughout the year (Hirano et al., 2007, 2009; Jauhiainen et al., 2008, 2005; Suzuki et al., 1999). Drainage and vegetation replacement therefore remove the two fundamental prerequisites required for peat accumulation and carbon storage (Delft Hydraulics, 2006; Hooijer et al*.* 2010; Jauhiainen et al., 2010). OP cultivation requires drainage depths of between 0.6 and 0.85 m (Henson & Dolmat, 2003; JRC, 2010; Singh, 2008) but in practice, these often exceed 1 m and even deeper drainage has been observed in the field (Delft Hydraulics, 2006; Hooijer et al. 2010, 2011). Although OP cultivation is possible at shallower drainage depths, palm productivity is reduced significantly and is less economically viable (Alterra, 2008).

The drainage of previously saturated peat immediately initiates the subsidence of the peatland surface and leads to a reduction in peat volume (Couwenberg et al., 2010; Delft Hydraulics, 2006; Hooijer et al*.* 2010; Wetlands International, 2009a; Wösten, Ismail & van Wijk, 1997). Subsidence is a function of the processes of peat consolidation, shrinkage, and compaction, and the decomposition (oxidation) of previously water saturated peat under aerobic conditions (Alterra, 2008; Delft Hydraulics, 2006; Hooijer et al*.* 2010; 2011; Wösten et al., 1997). Subsidence rates are rapid in the first one to two years following drainage, as the peat consolidates owing to increased overburden resulting from a loss of buoyancy (Hooijer et al., 2011) and can result in initial subsidence rates of more than 0.5 m yr^{-1} (Hooijer et al., 2011; Wösten et al., 1997). Following this primary stage of subsidence, a secondary phase of irreversible shrinkage and compaction of the peat together with rapid rates of peat decomposition leads to a slower but constant rate of subsidence (Hooijer et al., 2011; Wösten et al., 1997). The processes of consolidation, shrinkage, and compaction are entirely physical, and no carbon is lost, but peat bulk density (and carbon concentration) increases with time since drainage (Delft Hydraulics, 2006; Hooijer et al*.* 2010, 2011; Wösten et al., 1997). Use of machinery for drainage operations and consolidation of the peat before OP planting takes place further increases the bulk density of the surface peat. Although not considered here, losses of surface peat due to combustion during fire may also contribute to observed subsidence rates (Hooijer et al., 2010; Wösten et al., 1997).

The exposure of previously saturated peat to aerobic conditions leads to the rapid transfer of historically accumulated and previously stable carbon to the atmosphere (Delft Hydraulics, 2006; Hooijer et al., 2010, 2011; Renewable Fuels Agency, 2010). Recent studies show that OP litter inputs are quickly decomposed under aerobic conditions and unable to compensate for rapid rates of peat decomposition following drainage (Germer & Sauerborn, 2008). Peat CO_2 emissions are generally highest in the initial stages of drainage, owing to the rapid decomposition of a limited pool of labile carbon, but may decline over time as the relative amount of recalcitrant carbon compounds increases (Alterra, 2008; Couwenberg et al., 2010; Hooijer et al., 2011). Increases in peat temperature following drainage, however, particularly in the early stages of plantation establishment (or following replanting) before a closed canopy develops, along with applications of nitrogen fertilizers, may counterbalance this effect and maintain high rates of CO_2 emission (Hooijer et al., 2011; Jauhiainen et al., 2011; Murdiyarso et al., 2010). In order to sustain OP production, regular maintenance of the drainage channels is required as subsidence brings the water level back towards the peat surface (Delft Hydraulics, 2006; Hooijer et al. 2010; JRC, 2010). The cultivation of OP on tropical peat will, therefore, lead to the inevitable loss of peat with emission of CO_2 to the atmosphere (Fig. 6), the ultimate magnitude and timing of which will depend on peat depth and OP cultivation intensity (e.g., drainage and fertilization; Fargione et al., 2008; Renewable Fuels Agency, 2010). Before all of the peat ultimately disappears, however, the water table may reach the drainage base, at which point it becomes impossible to remove any more water; this results in flooding and brings an end to all plantation use. Furthermore, in addition to to the direct peat CO_2 emissions from OP plantations, losses can also occur from adjacent forested areas where the water table is lowered because of drainage in the plantation (Hooijer et al., 2011). These indirect emissions should also be attributed to OP plantations.

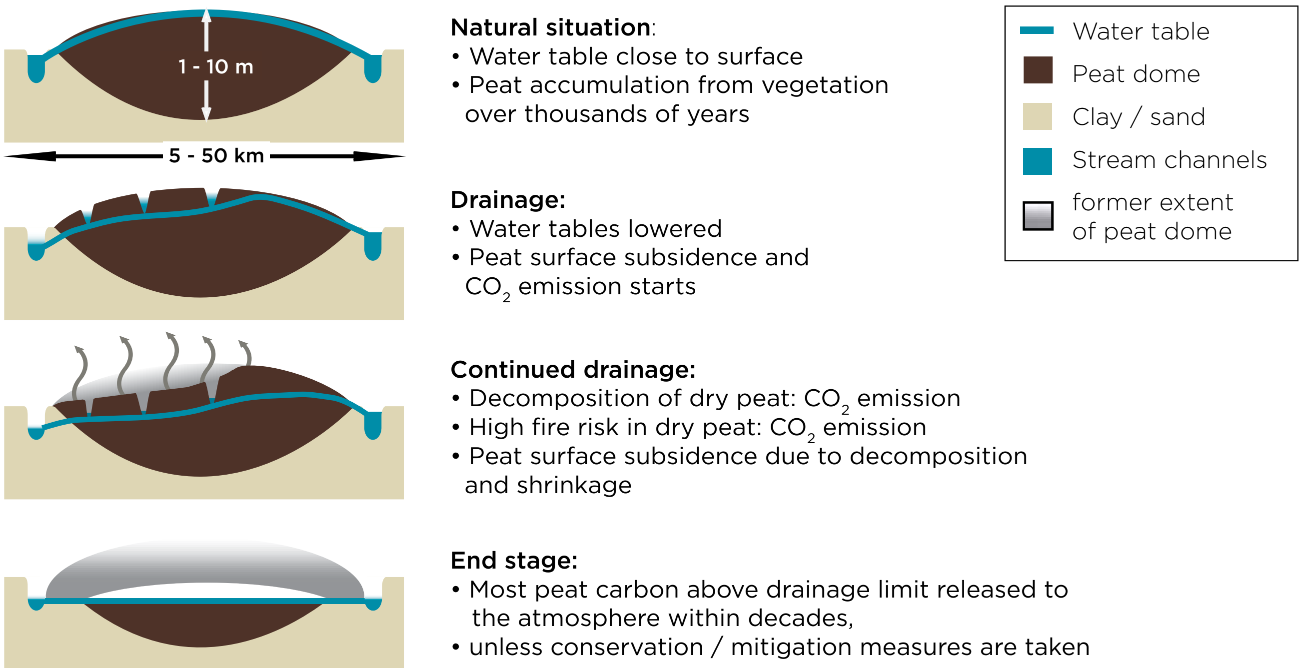

Figure 6: Schematic illustration of drainage effects on a peatland dome (modified from Delft Hydraulics, 2006)

In contrast to the large losses of CO_2, CH_4 emissions are reduced following drainage owing to increased rates of methanotrophy in the larger volume of aerated surface peat (Couwenberg et al., 2010; Jauhiainen et al., 2008; Melling et al., 2005b). CH_4 emissions may be high, however, from water surfaces in the network of ditches and canals required to drain plantations (Jauhiainen, unpublished data). Accelerated rates of nitrogen mineralization under drained aerobic conditions, together with high rates of fertilizer application, may contribute to high rates of N_2O emission (see section 2.5.3; Jauhiainen et al., 2011; Melling et al., 2007a; Murdiyarso et al., 2010).

4 CARBON CYCLE TERMINOLOGY AND GHG MEASUREMENT METHODOLOGIES

4.1 Terminology

In terms of carbon exchange processes (fluxes) in peat swamp forests, it is critical to define the appropriate carbon cycle terminology. According to Alterra (2008), much of the recent confusion surrounding carbon emissions from OP plantations on tropical peatland results from inadequate understanding of carbon cycle processes and the components of the peatland carbon balance captured by various measurement techniques. These misunderstandings have resulted in the misinterpretation of results presented by empirical studies. A study by Melling et al. (2005b), for example, which reported only on peat surface CO_2 emissions at a drainage affected peat swamp forest and at OP and sago palm plantations, has frequently been misinterpreted as representing the total net ecosystem-atmosphere CO_2 exchange. Subsequently, the erroneous conclusion has been made that large-scale conversion of natural peat swamp forest to OP plantation is beneficial in reducing net CO_2 emissions into the atmosphere (e.g., Tan, Lee, Mohamed, & Bhatia, 2009; Yew, Sundram & Basiron, 2010). This is based on the wrong assumption that the measurements of Melling et al. (2005b) included both peat surface emissions and photosynthetic uptake of CO_2 by vegetation. For example, the large amount of CO_2 emitted from the floor of an undrained peat swamp forest is likely to be mostly or completely reabsorbed by the vegetation, making the ecosystem CO_2 neutral or slightly negative (i.e., excess biomass production accumulates as peat). On the other hand, degraded peat swamp forest or plantation on drained peat may appear to be releasing similar quantities of CO_2 from the peat surface to an undrained forest, but a greatly reduced vegetation canopy will not be sequestering as much CO_2, so these systems will be CO_2-positive — i.e., net CO_2 emitters.

The net ecosystem carbon balance (NECB) of a peatland ecosystem consists of gaseous surface-to-atmosphere exchanges of CO_2 and CH_4, and fluvial losses of DOC and POC. The residual, after accounting for these losses and gains, manifests as increments to or losses from the forest biomass and peat carbon pool over time. The net ecosystem exchange of CO_2 (NEE) is the balance between the opposing fluxes of CO_2 sequestration in photosynthesis and release via respiration (Reichstein et al., 2005). Ecosystem respiration is the combined loss of CO_2 to the atmosphere from all the various respiring elements of the ecosystem (from above- and below-ground vegetation and associated mycorrhizae, other soil fungi. and bacteria); ecosystem respiration, therefore, represents the sum of autotrophic and heterotrophic respiration (Reichstein et al., 2005). Peat respiration is the total amount of CO_2 emitted at the peat surface, and is the combination of root and root-associated mycorrhizal (autotrophic) respiration and the respiration of heterotrophic macro- and microorganisms involved in the decomposition of organic matter (Jauhiainen et al., 2010). Root respiration is CO_2 released from the root zone (the rhizosphere) as a result of the metabolism of roots and root-associated mycorrhizal fungi, while peat heterotrophic respiration CO_2 emissions are from decomposition of organic matter (Alterra, 2008; Jauhiainen et al., in review, 2005, 2010). As all other CO_2 losses are derived from recent photosynthates, it is only carbon released via the decomposition of organic matter (fresh plant litter and peat) resulting in emission of CO_2 and CH_4 into the atmosphere (Kuzyakov, 2006), along with the release of DOC and POC into waterways, that contribute to net losses from the peat carbon pool (Moore et al., 2011). In practice, however, it is difficult to separate autotrophic from heterotrophic emissions in peat because of the problem of obtaining measurements under truly root-free conditions and also because organic matter decomposition in vegetated peatlands may also involve microorganisms that connect with plant roots (mycorrhizae).

The net ecosystem-atmosphere exchange of CO_2 (NEE) is the largest component of the carbon balance in tropical peat swamp forest and is essential to the long-term stability of peatland carbon stocks (Hirano et al., 2007). Where photosynthetic uptake of CO_2 exceeds all respiratory losses of CO_2 (denoted by negative values of NEE) there will be a surplus of carbon in organic matter within the system, preserved in biomass of live organisms or in dead organic matter, which, over time, forms peat. Peatland degradation of any form (logging, drainage etc.) disrupts the ecosystem's CO_2 balance; NEE becomes positive, reflecting a net transfer of carbon from the system to the atmosphere, as a consequence of both reduced carbon inputs (reduced biomass and photosynthetic capacity) and increased carbon outputs (principally from increased heterotrophic respiratory losses, i.e., decomposition of peat and litter).

The emission or consumption of CH_4 represents the balance between methanogenesis (CH_4 production) under anoxic conditions and methanotrophy (CH_4 consumption) within the oxic peat profile (when present). Additionally, CH_4 may also be emitted to the atmosphere via ebullition (bubbling) or via plant aerenchyma tissues (e.g. pneumataphore root structures in some peat swamp trees; Couwenberg et al., 2010). All studies to date indicate that the net volumes of methane exchanged between the peat surface and the atmosphere are relatively low in tropical peatlands, i.e., less than 10% $CO_{2\text{-eq}}$, and that drainage further reduces the peat CH_4 emission potential (Jauhiainen et al., 2011; Melling et al., 2005a).

4.2 Methods of assessing greenhouse gas emissions

Various methods have been used for estimating carbon and GHG fluxes and budgets in tropical peatlands. In general, these can be classified as: (a) direct measurements of gaseous (and fluvial) fluxes that are applicable to all three biogenic greenhouse gases (CO_2, CH_4 and N_2O), and (b) estimates based on peat subsidence rates (only relevant to carbon loss from drained peatlands). This section provides an overview of these measurement techniques. It is important to note that each technique has its advantages and disadvantages, largely relating to the spatial and temporal scales of measurement. Additionally, it is critical to have a clear understanding of exactly which components of the carbon and GHG budgets are measured by each method and, perhaps more importantly, which components are not measured or cannot be adequately differentiated. It is worth restating that it is only the carbon released from decomposition of historically accumulated peat that is of relevance to global carbon emissions and anthropogenic climate change (Couwenberg et al., 2010; Kuzyakov, 2006).

4.2.1 FLUX CHAMBERS

The most widely used method of assessing surface-to-atmosphere GHG fluxes in tropical peatlands is the closed chamber (or enclosure) technique (Figure 7; Ali et al., 2006; Hirano et al., 2007, 2009; Jauhiainen et al., 2011, 2005; Melling et al., 2005a, b, 2007a, b; and others reviewed in Couwenberg et al., 2010). Chamber methods provide data on surface-to-atmosphere gaseous flux exchange from small, well-defined areas at specific points in time that are scaled up to provide time-integrated GHG budgets for longer time periods and larger areas. While a number of variants on this methodology exist (e.g., chamber size and design and whether chambers are static or dynamic, clear or opaque), the basic measurement principle involves creating a sealed volume over an area considered to be representative of surface conditions (Denmead, 2008). In practice, an airtight vessel is placed directly onto the peat surface (usually using a fixed collar) to prevent leakage and allow gas concentrations to increase in the chamber headspace over time. Gas concentrations are quantified either by the static method, in which samples are collected in syringes at specific intervals for subsequent laboratory analysis, or the dynamic method, in which air is circulated from the chamber headspace to a portable gas analyser to provide instantaneous readings in the field (Denmead, 2008). Flux rates are determined subsequently by linear regression of the change in gas concentration as a function of incubation time. Readings are typically rejected when the change in gas concentration is non-linear (indicating leaks or soil disturbance), or when the coefficient of gas concentration linearity (r^2) is less than 0.9 in the samples included in the analysis (Baird, Holden, & Chapman, 2009). Thus, appropriate use of static closed chamber method requires an absolute minimum of three gas subsamples to be taken during incubation and included in the calculation of each flux estimate. Closed chamber incubation time must be adjusted so that the analytical system being used is able to determine GHG flux with the required level of accuracy — i.e., slow gas exchange rates require longer incubation periods. The instantaneous readings provided by dynamic systems, which use high gas sampling frequency for flux determination, are therefore advantageous in enabling real-time data quality assessment.

Figure 7: Closed chamber measurements being made in the field; large static chamber (left) and dynamic chamber and CO_2 analyzer (right; images: J. Jauhiainen).

Closed chambers may be either clear (transparent) or dark (opaque). The use of clear chambers allows photosynthetic uptake of CO_2 by the vegetation to be accounted for and can therefore be used to measure the net ecosystem CO_2 exchange with the atmosphere. This technique can, however, only be applied in low-growing vegetation and is of no use in forested peatland. A further challenge caused by the use of sealed, clear closed chambers in a hot tropical climate is the maintenance of a stable temperature during vegetation light- and dark-reaction measurements. The use of dark closed chambers excludes the photosynthetic capture of CO_2 by vegetation. When using dark chambers, vegetation must be absent from the enclosed chamber area in order to ensure that contributions from plant respiration are not included in the analyzed CO_2 concentration. Practically all closed chamber measurements in tropical peatlands have employed the dark chamber technique to assess peat surface CO_2 emissions, because the stature of the forest or plantation vegetation means that the small volume measurement chambers are practical only for capturing CO_2 emitted at the peat surface (i.e., soil-to-atmosphere gas exchange only, excluding the above-ground vegetation processes of photosynthesis and respiration). In situations where the vegetation stature is low (e.g., at some tropical agricultural sites), it has been possible to use a combination of clear (transparent) and dark chambers to obtain estimates of the net ecosystem exchange of CO_2 (i.e., including the light and dark CO_2 exchange processes in the enclosed vegetation).

The majority of investigations using the dark closed chamber technique in tropical peatlands provide measurements of undifferentiated (total) peat respiration. From such data, it is not possible to quantify the CO_2 emission portion attributable to peat or litter decomposition (heterotrophic respiration) from emissions from vegetation root respiration (autotrophic respiration), either because these studies provide no information on the volume of roots at the monitoring sites and/or because there is insufficient reporting of the measurement conditions and of field methodologies to exclude CO_2 contributions from subsurface vegetation respiration. As root respiration (respiration of root systems, recent root litter, and root exudates) is derived from recent photosynthates that do not contribute to net carbon losses from the peat carbon pool, site comparisons of net peat CO_2 emissions using dark chambers are largely ineffective owing to variations in root biomass and associated autotrophic respiration rates (Couwenberg et al., 2010; Hooijer et al., 2010; Jauhiainen et al., in review, 2008, 2005). Closed chamber measurements of peat surface CH_4 and N_2O are generally less complex than for CO_2, as emissions are derived from organic matter decomposition only, but care is required to avoid disturbances that may artificially trigger episodic ebullition of these gases (Baird et al. 2009; Couwenberg et al., 2010; Jauhiainen et al., 2011; Wetlands International, 2009a).

Closed chambers have been used to quantify peat surface emissions of all three biogenic GHGs in a range of tropical peatlands (Ali et al., 2006; Chimner, 2004; Chimner & Ewel, 2004; Darung, 2004; Forest Research Institute Malaysia, 2008; Furukawa et al., 2005; Hadi et al., 2005; Hadi, Inubushi, Purnomo, Furukawa, & Tsuruta, 2002; Hadi et al., 2000; Hirano et al., 2007, 2009; Inubushi et al., 2003; Inubushi & Hadi, 2007; Inubushi et al., 2005; Ismail, Zulkefli, Salma, Jamaludin, & Mohamad Hanif, 2008; Jauhiainen et al., 2008, 2005; Jauhiainen, Vasander, Heikkinen, & Marikainen, 2002; Jauhiainen et al., 2004; Kyuma, Kaneko, Zahari, & Ambak, 1992; Melling et al., 2007a, Melling, Goh & Hatano, 2006; Melling et al., 2005a, b, 2007b; Murayama & Bakar, 1996; Rumbang, Radjagukguk & Projitno, 2008; Southeast Asia Regional Committee for START, 2001; Suzuki et al., 1999; Takakai et al., 2005, 2006; Ueda et al., 2000; Vien, Phuong, & Jauhiainen, 2008). It should be noted, however, that most of these studies have not claimed to address the quantification of gaseous emissions from peat decomposition only, although some of these data have been used subsequently for this purpose.

Several techniques have been developed to differentiate total peat CO_2 emissions into autotrophic and heterotrophic components. These include isotopic techniques and trenching. According to Kuzyakov (2006) and Couwenberg et al., (2010), isotopic techniques, based on analysis of stable or radiocarbon isotopes, are valid only under controlled laboratory conditions and are often prohibitively expensive and generally imprecise, while methods that attempt to exclude root respiration in the field are often destructive or may alter system dynamics in ways that prevent robust estimates of heterotrophic CO_2 emissions. The most common, although not widely used, method of differentiating root and peat emissions in tropical peatlands is trenching, whereby roots are severed prior to the start of gas flux monitoring by closed chambers (Jauhiainen et al., in review; Melling et al., 2007b); subsequently, the peat is cut in order to prevent root ingrowth. Severed roots may continue to respire for months after trenching; trenching may also alter thermal and hydrological properties of the peat and remove the rhizosphere priming effect (whereby the presence of roots and root exudates stimulate heterotrophic processes). Thus, peat CO_2 emissions are likely underestimated over the long term by this approach (Couwenberg et al., 2010; Jauhiainen et al., 2011b; Wetlands International, 2009a). According to Couwenberg et al., (2010), it remains doubtful that any study has truly managed to report differentiated decomposition CO_2 emissions from peat, although a recent study by Jauhiainen et al. (in review) appears to have achieved this by combining the trenching approach with measurements made along transects at variable distances from evenly spaced plantation trees, i.e., both within and beyond the principal tree rooting zone, enabling assessment of essentially root-free respiration from the peat surface.

Other limitations of closed chamber estimates may include: long chamber incubation times that may alter temperature and pressure conditions that, in turn, may influence emission rates from the soil; alterations to the microclimate within the chamber headspace; and the methods used for extrapolating infrequent small-scale measurements to larger spatial and temporal scales (Denmead, 2008). Spatial and temporal up-scaling from closed chamber estimates typically involves using averages of replicate measurements, assumed to represent the full range of spatial variability present in the ecosystem, multiplied by area and time. This implies that the full range of spatial variability is captured by the measurements, when in reality it is not due to the inherent spatial heterogeneity of peat substrates, and that the relative area of each 'habitat' type is accurately known (Becker et al., 2008; Jauhiainen et al., 2005). Sensitivity analyses provided by Jauhiainen et al. (2005) showed that annual estimates of peat surface emissions in intact peat swamp forest depended strongly on assumptions relating to peat surface microtopography and the relative proportions of hummocks and hollows (i.e., areas of relative dryness and wetness).

In terms of temporal upscaling, GHG emissions from the peat surface often show marked diurnal and seasonal cycles (Ali et al., 2006; Hirano et al., 2009), and there may be lags between biological gas production within the peat profile and emissions measured at the surface (Denmead, 2008). Moreover, most measurements of GHG emissions from tropical peatlands have been made during the day over relatively short time intervals and at only a few times during the year (e.g., Melling et al., 2005a, b, 2007a, b; Murayama & Bakar, 1996). Closed chamber measurements biased towards one part of the diurnal (or seasonal) cycle are inadequate and may prove misleading when extrapolated in time (Denmead, 2008; Jauhiainen et al., 2011). To fully account for diurnal effects, closed chamber measurements should, ideally, be conducted over complete diurnal cycles (Denmead, 2008). Furthermore, adequate accounting for seasonal and inter-annual variability using closed chamber techniques requires numerous, high frequency measurements to be made over long periods of time (Denmead, 2008; Jauhiainen et al., 2011) or, although not yet generally used for tropical peatlands, the use of a series of automated chamber systems capable of providing continuous GHG flux measurements (Denmead, 2008; Hirano et al., 2009; Wang, Dalal, Reeves, Butterbach-Bahl, & Kiese, 2011). In general, the majority of studies using closed chamber methods for tropical peatlands have not been based on sufficient numbers of replicates over sufficient length of time to provide statistically robust flux values or uncertainty ranges.

4.2.2 EDDY COVARIANCE

Eddy covariance (EC) measurements are based on micrometeorological (aerodynamic) theory; a full description of the technique is beyond the scope of this review (see: Baldocchi, 2003; Law & Verma, 2004). In general terms, EC uses fast response (typically 20 Hz) sensors mounted on a tower up to a number of meters above the vegetation canopy to sample the vertical component of atmospheric turbulence (using a sonic anemometer), and the concentration of the atmospheric scalar (e.g., greenhouse gas, water vapor) of interest (using infrared gas analysis for CO_2 or quantum cascade lasers for CH_4 and N_2O; Fig. 8). Fluxes are computed as the mean covariance between the vertical wind speed and the concentration of the relevant GHG. The EC technique is widely considered as the most appropriate method of quantifying ecosystem-atmosphere greenhouse gas budgets (e.g. Baldocchi, 2003; Laine et al., 2006). It provides direct measurements of net ecosystem-atmosphere GHG exchange across the vegetation-soil-atmosphere interface (Baldocchi, 2003). In contrast to closed chambers, EC is able to provide continuous, multiyear, whole ecosystem gaseous flux measurements (e.g., including the photosynthesis and respiration of the forest vegetation) from a relatively large (hectares to km^2) source area or "flux footprint" (Baldocchi, 2003). Following a number of corrections, largely related to atmospheric conditions and limitations inherent to EC measurement systems, surface-atmosphere fluxes are obtained as 30-minute to hourly averages that can be integrated over time, providing daily, seasonal, and annual budgets (Aubinet et al., 2000). It should be noted that while the aim of EC is to provide continuous measurements, some level of data gap filling is always required due to unavoidable data loss (Baldocchi, 2003; Moffat et al., 2007; Reichstein et al., 2005).

Figure 8: An eddy covariance flux tower measuring surface-atmosphere exchanges of CO_2 water and energy at a restored fen peatland in the United Kingdom (left). Eddy covariance instrumentation, consisting of a sonic anemometer, an infrared gas analyzer, and sensors for measuring photosynthetically active radiation, air temperature and relative humidity (right). (Images: R. Morrison).

The EC technique is in widespread use across the globe as the primary tool of the global FLUXNET community (Baldocchi et al., 2001; FLUXNET, 2011). In addition to the EC system, EC stations are generally equipped with a suite of meteorological sensors (e.g., radiation, temperature, precipitation, humidity, and soil moisture content) that facilitate analyses of whole ecosystem responses to diurnal, seasonal, and interannual variability in environmental conditions. When deployed within the FLUXNET framework, EC enables a previously unprecedented understanding of the functioning of the carbon dynamics of the terrestrial biosphere (Baldocchi et al., 2001). Historically, the method has been widely used to measure the net ecosystem exchange of CO_2 and energy fluxes; however, recent advances in instrumentation are now enabling EC measurements of CH_4 and N_2O (e.g., di Marco, Skiba, Weston, Hargreaves, & Fowler, 2004; Rinne et al., 2007). Despite the widespread adoption of EC and a growing number of studies in northern peatlands (e.g., Hendricks, van Huissteden, Dolman, & van der Molen, 2007; Lloyd, 2006; Roulet et al., 2007), there are few published EC studies addressing CO_2 balances from tropical peat swamp forests (Hirano et al., 2007, 2009; Suzuki et al. 1999), and none that have reported on emissions from OP plantations on peatlands.

In the case of CO_2, the EC technique provides a measurement of the NEE of CO_2 (the balance between photosynthetic uptake and losses via ecosystem respiration). Although it is possible to partition NEE into GPP and total ecosystem respiration using modeling approaches (Desai et al., 2007; Lasslop et al., 2010; Reichstein et al., 2005), it is not possible to partition ecosystem respiration into its autotrophic and heterotrophic components on the basis of EC measurements alone (Couwenberg et al., 2010; Wetlands International, 2009a). Partitioning of net peat CO_2 emissions would require additional use of closed chambers and/or detailed accounting of biomass, litter and peat carbon stocks (Couwenberg et al., 2010; Wetlands International, 2009a). It is also important to note that EC provides a spatial average of GHG emissions from within the flux footprint, and does not provide information on small-scale processes operating at lower spatial scales, which could lead to erroneous conclusions where emissions "hot spots" exist (Becker et al., 2009; Laine et al., 2006; Teh et al., 2011). Moreover, the footprint of EC measurements varies according to wind direction and atmospheric conditions, and care is required in interpreting results where heterogeneous surface conditions exist and in relating large-scale flux measurements to spatially variable drivers such as drainage depth. Further logistical considerations include the expense and technical problems associated with installing a tower and providing a continuous energy source.

4.2.3 MEASUREMENTS OF DOC AND POC

Losses of DOC and POC from peatlands are not captured by closed chamber or eddy covariance techniques, and estimation requires measurements of both DOC and POC concentrations and fluvial discharge rates (Moore et al., 2011). Loss rates are then estimated by multiplication of fluvial carbon concentrations with flow rates. Thus far, estimates of DOC and POC losses from tropical peatlands have been based on measurements from rivers draining peat-covered catchments and are applicable only at large spatial and temporal scales (Baum et al., 2007; Moore et al., 2010; Fig. 9). Measurements of fluvial carbon discharge from catchments containing OP plantations are required.

Figure 9: Blackwater stream draining a peatland in Central Kalimantan, Indonesia (Image: S. Moore).

4.2.4 SUBSIDENCE MONITORING

In addition to direct flux measurements, it is also possible to estimate net carbon emissions from drained peatlands on the basis of peat surface subsidence rates (Couwenberg et al., 2010; Delft Hydraulics, 2006; Hooijer et al. 2010, 2011; Wösten et al., 1997). Peat carbon losses are (or should be) reported as $CO_{2\text{-eq}}$ (carbon dioxide equivalents); however, because most of the emissions are gaseous CO_2 losses, the subsidence method does not measure CO_2 emission *per se*. The subsidence method integrates total carbon loss and thus is insensitive to the differing forms of carbon (CO_2, CH_4, POC, DOC); these need to be studied by other techniques if information on the proportional contribution of each carbon form is required, e.g., in order to calculate GWP contributions correctly. Data on carbon loss derived from subsidence monitoring data are generally considered to be more reliable for estimating carbon losses from drained peat than those obtained from closed chambers, because they are capable of providing a time-integrated measure of the net carbon balance of the peat. The relatively long integration period required for successful application of the subsidence method means, however, that it is not sufficiently sensitive to detect changes in carbon dynamics over short time scales — i.e, the impact of possible diurnal or even seasonal differences on carbon loss cannot be measured. For experimental studies on carbon dynamics (such as studies on the impact of fertilization on peat decomposition), the method requires long time periods and large-scale experiments across reference areas.

Figure 10: Subsidence pole inserted in peatland in Johor, peninsular Malaysia. The pole was inserted beside an oil palm plantation in 1978 and at the time of this photograph (2007), 2.3 m of subsidence had occurred (the human "measuring stick," Dr. Chris Banks, is 2 m tall). (Image: J. Jauhiainen).

Subsidence monitoring exploits measurements of changes in peat surface position over time, as a measure of changes in peat thickness (Fig. 10). As subsidence is a function of both physical processes of compaction and consolidation and the biological process of peat decomposition (oxidation), the relative contribution of peat decomposition to the overall subsidence must be determined using peat bulk density profiles and measurements of peat carbon concentration obtained both before and after drainage (Couwenberg et al., 2010; Hooijer et al., 2011). For example, if the peatland surface and volume are reduced by half in any given time, and peat bulk density has doubled, then all subsidence can be attributed to mechanical processes; if all other factors remain constant and peat bulk density has not changed, then all subsidence can be said to be from peat decomposition (Hooijer et al., 2011). Published estimates of the relative contribution of peat decomposition to overall subsidence rates vary significantly; values ranging from 40% to 100% have been reported for peatlands globally (Couwenberg et al., 2010; Hooijer et al., 2011; Murayama & Bakar, 1996; Murdiyarso et al., 2010). It should, however, be noted that the lower numbers in this range apply to temperate climates, and that the highest decomposition contributions are reported and expected for tropical climates as peat decomposition is highly temperature-dependent (Stephens, Allen & Chen, 1984). According to Hooijer et al. (2011), the percentage of subsidence attributed to decomposition increases in the years following drainage. This implies that carbon emissions from peat decomposition are highest in the years immediately after drainage, but the relative contribution of heterotrophic peat decomposition to overall subsidence rates increases with time. The very fact that drained peat eventually disappears from the landscape confirms that decomposition must ultimately account for 100% of observed subsidence (Hooijer et al., 2011). Previously, overestimation of the compaction and shrinkage component in the overall long-term subsidence process, largely due to inadequate or insufficient carbon content and bulk density measurements, has led to underestimation of the carbon emissions arising from peat decomposition and oversimplification of the physical subsidence process (e.g., Hergoualc'h & Verchot, 2011; Wösten et al., 1997).

The key limitation of subsidence monitoring to quantify carbon loss is that subsidence is a slow process and may require monitoring over a number of years before estimates of carbon loss can be obtained. However, this period can sometimes be shortened to two years or even one if measurements meet the highest standards, are conducted in sufficiently large numbers at tens or even hundreds of locations, and coincide with a period of relatively "normal" weather conditions (Hooijer et al., 2011). A long observation period increases the risk of occasional disturbances at individual poles and thus the loss of subsequent data for that measurement location, so trade-offs often need to be made among monitoring duration, number of monitoring points, and monitoring quality. Additional problems relate to the availability of high-quality bulk density data for complete peat profiles obtained both before and during multiple time steps after drainage, as well as the assumption that drainage is the only process driving subsidence, which does not explicitly consider the interactive effects of peat depth and type, land use history, temperature, and fertilization (Murdiyarso et al., 2010) or micro-scale variations in peat characteristics and topography. In addition, confidence limits or reliability analyses have not been provided for most subsidence data. Although subsidence is typically monitored using poles anchored into the substrate underlying the peat (Fig. 10), remote sensing techniques can be used to monitor subsidence over large spatial scales (Ballhorn, Siegert, Mason, & Limin, 2009). Providing the rate of peat subsidence and the proportion attributable to decomposition can be adequately determined, the combination of satellite data and modeling presents a promising and accurate means of assessing peat carbon emissions at both local and regional scales (Couwenberg et al., 2010).

4.3 Methods conclusion

From the above review, it is clear that while each method of assessing carbon and GHG emissions from drained tropical peatlands has its relative advantages, no method alone is capable of assessing the full peatland GHG balance. Most of the criticisms of tropical GHG flux studies to date have focused on studies using the closed chamber measurement technique since these have been the most numerous, but all the methods identified above need modification based on the particular characteristics of the tropical peatland environment. In terms of direct gaseous flux measurements, Laine et al. (2006) suggest that EC is the measurement technique of choice for research relating to long-term ecosystem-scale GHG budgets, while closed chamber techniques are better suited to developing environmental response functions and for assessments of the spatial variability in carbon containment and other GHG emissions from peat. Subsidence monitoring can provide an accurate, time-integrated measure of total carbon loss from peat soils following drainage, but often needs monitoring over longer time periods and requires accurate measurements of peat bulk density profiles for estimating the fraction of subsidence that is attributable to peat decomposition.

In the ideal case, emissions from tropical peatlands should be monitored using a combination of all methods: (a) EC to provide continuous gaseous flux measurements at the ecosystem scale, subsidence monitoring to assess total carbon loss from peat soils; (b) closed chamber measurements (ideally using automated chamber systems) for assessing smaller-scale emissions dynamics for various GHGs and spatially explicit responses of peat surface GHG flux dynamics to local environmental conditions (e.g., location on peat dome, peat type and depth, depth of groundwater table, time since drainage, fertilization, etc.); and (c) measurements of TOC (DOC and POC) to assess lateral movements of carbon and subsidence monitoring to assess peat-only carbon emissions. Additionally, a coordinated measurement framework is required to monitor and compare the scale of carbon emissions from peat swamp forests and peatland under a range of other land uses, including OP plantation. The temporal transition from native peat swamp forest to OP plantation should be captured in order to account for variations in emissions with time since land use conversion, whilst the spatial transition from undrained to drainage-affected forest should be covered in order to account for the impact that plantation drainage has on enhanced peat emissions several kilometers beyond the plantation drainage system (Hooijer et al*.,* 2011). Full accounting of the regional variability in GHG emissions in response to large-scale biophysical variations (e.g., rainfall regimes) requires measurements at a number of sites across the peatlands of the Southeast Asian region.

5 REVIEW OF PUBLICATIONS ON EMISSIONS FROM OIL PALM PLANTATIONS

This review indentifies a number of recent publications addressing GHG emissions from OP plantations on tropical peat (e.g., Danielsen et al., 2009; Fargione et al., 2008; Germer & Sauerborn, 2008; JRC, 2010; Koh et al., 2011; Murdiyarso et al., 2010; Reijnders & Huijbregts, 2008; Wicke, Dornburg, Jungiger, & Faaij, 2008). The key features and results of these studies are summarized in Table 2. The publications reviewed differ in scope, some addressing GHG emissions for the entire OP biofuel production-transport-consumption chain (Fargione et al., 2008; Reijnders & Huijbregts, 2008; Wicke et al. 2008), others focusing solely on land use related emissions (Germer & Sauerborn, 2008; Koh et al., 2011; Murdiyarso et al., 2011). In addition to estimating emissions following land use conversion of tropical peat swamp forest to OP plantation, a number of studies also provide estimates of emissions from ecosystems other than peatlands, such as: rain forests on mineral soils (Fargione et al., 2008; Wicke et al., 2008), degraded lands (Danielsen et al., 2009; Wicke et al., 2008), and anthropogenic grass-lands (Germer & Sauerborn, 2008). Studies also differ in how the impacts of OP plantation developments are quantified, some providing estimates of GHG emissions over the complete OP production cycle (Germer & Sauerborn, 2008), others also providing annualized emissions (Murdiyarso et al., 2010), and some focusing on the biofuel carbon debt or carbon payback time required to compensate for biofuels derived from OP feed-stocks (Fargione et al., 2008; Gibbs et al., 2008). Despite these differences, however, the conclusion from all studies is that whilst carbon losses from biomass replacement and land clearance are considerable, it is the large and sustained CO_2 emissions from drained peat that contribute most to overall emissions and carbon debts. According to Danielsen et al. (2008), Fargione et al. (2008), and Gibbs et al. (2008), for example, the conversion of tropical peat swamp forest for OP production results in biofuel carbon debts that would require between 420 and 900 years to repay. It is also noteworthy that studies considering OP plantation establishment on grasslands or degraded lands have estimated that OP produced on these areas could lead to much shorter carbon payback times (Danielsen et al., 2009) and higher biomass carbon storage in OP relative to the former ecosystems (Germer & Sauerborn, 2008).

Table 2 (next page): Summary of key components and findings of recent studies aiming to quantify ecosystem level greenhouse gas emissions from oil palm production systems in Southeast Asia; Only studies considering emissions from the entire ecosystem have been included. Studies providing emissions estimates for peat surface emissions are considered in more detail in the main text and in Table 3. Negative values denote a net sink.

REFERENCE	DESCRIPTION	KEY FINDINGS
Germer & Sauerborn (2007)	Literature review and analysis aiming to quantify greenhouse gas emissions from OP plantations on tropical rain forests (on mineral soils), peatlands and anthropogenic grasslands. Considered changes in biomass carbon storage, emissions from land clearance (including fire) and emissions from soils.	Estimated net greenhouse gas balance over 25-year production cycle: -134 ± 36 Mg $CO_{2\text{-eq}}$ ha^{-1} for degraded grasslands (i.e. net sink); 668 ± 372 Mg $CO_{2\text{-eq}}$ ha^{-1} for forest conversion on mineral soils; 1335 ± 690 Mg $CO_{2\text{-eq}}$ ha^{-1} for forest conversion on peatland.
Fargione et al. (2008)	Analysis aiming to quantify the biofuel carbon debt of various biofuel production systems. Considered OP production on lowland tropical rainforests on mineral soils in Southeast Asia. Considered land use and other components of the OP biofuel production chain. Emission estimates obtained from literature review.	Estimated emissions (over a 50-year period) and carbon biofuel debts of: 610 Mg $CO_{2\text{-eq}}$ ha^{-1} and carbon debt of 86 years for lowland forest; 3000 Mg $CO_{2\text{-eq}}$ ha^{-1} and carbon debt of 420 years for peatland; also considered emissions over 120 years for peats deeper than 3 m and estimated emissions of 6000 Mg $CO_{2\text{-eq}}$ ha^{-1} and biofuel carbon debt of 840 years.
Reijnders & Huijbregts (2008)	Assessment of gaseous carbon emissions from the OP production cycle in Southeast Asia. Focus is on OP plantations that replace tropical forest. Emissions estimated from literature review. Noted the scarcity of data on carbon emissions from burning and peat respiration and suggested emission values used may not adequately represent true emissions.	Estimated aboveground biomass carbon losses of 27.5 Mg CO_2 ha^{-1} yr^{-1} over a 25 year plantation life cycle and losses of CO_2 from peat at 36.7 to 55 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} over a 25-year plantation life cycle. Estimated CO_2 emissions of 1.5 to 5.8 Mg $CO_{2\text{-eq}}$ per ton of palm oil produced from converting forests on mineral soils, and 9 to 17 Mg $CO_{2\text{-eq}}$ per ton palm oil for conversion of forest on peatland.
Wicke et al. (2008)	Study aimed to quantify greenhouse gas emissions from palm oil production in Borneo for electricity generation in the Netherlands. Considered greenhouse gas emissions from conversion of degraded grasslands, forests on mineral soils and forests on peatlands. Compared to fossil energy production systems according to the 50-70% greenhouse gas emission reductions sustainability criteria of the Cramer Commission. Biomass and emissions data obtained from IPCC defaults and literature review.	Concluded that land use change is the most decisive factor in overall greenhouse gas emissions. OP plantations on degraded and well managed lands can be a net carbon sink. OP plantations on former rain forests on mineral soils and peatlands have such large greenhouse gas emissions that they will not be able to meet 50-70% GHG emissions reductions relative to fossil energy systems.
Danielsen et al. (2009)	Analysis aimed to quantify carbon payback times and biodiversity losses associated with OP plantation development for biofuel production. Considered OP plantation development on rainforests on mineral soils, peatland and Impereta grassland. Emissions estimated from the literature. Considered emissions from forest clearance (logging and burning), replacement of forest biomass, CO_2 emissions from drained peat soils. Also included emissions from non-land use related components of the OP biofuel chain.	Estimated carbon payback times of: 75 to 93 years for OP production on tropical forests (mineral soils); 692 years for OP production on peatland; 10 years for Imperata grasslands.
JRC (2010)	Report on iLUC modeling of biofuel feedstock expansion. IFPRI-MIRAGE model used to assess emissions from OP on tropical peatland.	Highlighted that the values used to estimate emissions from drained peatlands used for production of OP feedstocks are likely to be underestimated due to uncertainties in the area of peatland used for production, and the values used to represent CO_2 emissions from drained peat are based on values that do not account for the deep drainage required for OP.
Murdiyarso et al. (2010)	Analysis of ecosystem scale carbon loss from land use conversion of tropical peatland to OP and Sago plantations. Biomass and emissions data obtained from the literature. This study is unique in attempting to account for all ecosystem C fluxes using literature values. Peat respiration estimated by subtracting root emissions estimated using a physiological model calibrated for mineral soils. Balanced inputs and outputs to the peat carbon pool using estimates of carbon lost by land clearance using fire, above- and belowground biomass inputs and losses of fluvial carbon. Estimates of biomass inputs estimated from mineral soils; fluvial losses of carbon based on losses from northern peatlands.	Estimated that land use conversion of tropical forests would result in total carbon emissions of 59.4 ± 10.2 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} over a 25-year production cycle. Estimated 61.6% of CO_2 emissions are from peat; 25% of emissions are released by land clearance by fire.
Koh et al. (2011)	Remote sensing analysis aiming to assess biodiversity and C cycle impacts of OP plantation development in peninsular Malaysia, Sarawak and Borneo. Estimated carbon emissions resulting from land use conversion on the basis of calculations provided by Murdiyarso et al., (2010)	Identified 880,000 hectares of oil palm plantations on peat soils in Peninsular Malaysia, Sarawak and Borneo in 2010. The extent of oil palm plantations was underestimated, as only closed canopy plantations covering areas greater than 200 hectares were identified. Estimated total CO_2 losses.

The values used as the basis for estimating GHG emissions from drained peat soils in the studies reviewed above are summarized in Table 3. It was not clear how peat emissions were calculated in the studies of Gibbs et al. (2008) and Danielsen et al. (2009), so these have been omitted. In terms of CO_2, the values representing emissions from drained peat vary considerably, with the 19.2 Mg CO_2 ha^{-1} yr^{-1} estimate of Murdiyarso et al. (2010), which is also used by Koh et al. (2011), being nearly three times lower than the 55 and 57 Mg CO_2 ha^{-1} yr^{-1} values used by Fargione et al. (2008) and JRC (2010), respectively. Whilst two of the studies reviewed also include emissions of N_2O in their analyses (Germer & Sauerborn, 2008; Wicke et al., 2008), only Germer & Sauerborn (2008) considered emissions of all three biogenic GHGs. These authors estimated total peat surface emissions of 33 ± 16 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} resulting from emissions of CO_2 and N_2O, and consumption of CH_4 in the oxic peat profile. It is also significant that all studies consider emissions from drained peats to be constant over time, and do not consider the potential for interannual variations in emissions, or that peat CO_2 emissions in particular are likely to be highest in the early stages of the OP production life cycle (Hooijer et al., 2011) or following repeated drainage activities (JRC, 2010). Moreover, regional geographical differences in peat emissions in relation to key biophysical parameters such as precipitation or spatial differences across individual peat domes have yet to be explicitly considered, as have the conversion of degraded peat swamp forest or the development of OP plantation on existing agricultural land (Paoli et al., 2011).

Table 3: Values by various authors to estimate greenhouse gas emissions from drained peats under oil palm cultivation. Cells marked as NC indicate that the gas was not considered in the study. Negative values indicate a net greenhouse gas removal from the atmosphere. It was not clear how peat emissions were calculated in the studies of Gibbs et al. (2008) and Danielsen et al. (2009), so these studies have been omitted.

REFERENCE	EMISSIONS FROM DRAINED PEAT			
	CO_2 Mg CO_2 ha^{-1} yr^{-1}	CH_4 Mg CH_4 ha^{-1} yr^{-1}	N_2O kg N_2O-N ha^{-1} yr^{-1}	$CO_{2\text{-eq}}$ Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}
Fargione et al. (2008)	55	NC	NC	55
Germer & Sauerborn (2008)	31.4 ± 14.1	-0.2	4.1 ± 5.5	33 ± 16
Reijnders & Huijbregts (2008)	36.7 to 55	NC	NC	36.7 to 55
Wicke et al. (2008)	39	NC	8	42.7
JRC (2010)	57	NC	NC	57
Murdiyarso et al. (2010)	19.2	NC	NC	19.2
Koh et al. (2011)	19.2	NC	NC	19.2

The lower peat CO_2 emissions estimate of Murdiyarso et al. (2010), which is also used by Koh et al. (2011), can be explained by the use of the IPCC (2006) gain-loss approach to estimate peat-only emissions. In contrast to all other studies reviewed, which assumed that annual CO_2 emissions could be estimated using values obtained directly from the IPCC (2006) or published values (studies reporting measurements of emission values are reviewed in more detail below), Murdiyarso et al. (2010) balanced estimates of total peat surface CO_2 emissions measured using closed chamber techniques (Melling et al., 2005b; Murayama & Bakar, 1996) with above- and below-ground inputs of OP biomass (Henson & Dolmat, 2003; Lamade & Bouillet, 2005) and losses of carbon via root respiration (Henson & Dolmat, 2003; van Kraalingen, Breure & Spitters, 1998), combustion (Rieley & Page, 2008) and fluvial carbon loss (Holden, 2005). Methane was omitted from the analysis on the grounds that CH_4 emissions and consumption were negligible compared to high rates of CO_2 emission over a typical 25-year OP production cycle (Murdiyarso et al., 2010). Whilst the approach is promising, the study of Murdiyarso et al. (2010) highlights the dearth of data available for use in their calculations. Root respiration, for example, was estimated using a physiological model calibrated for mature OP grown on mineral soils (Henson & Dolmat, 2003; van Kraalingen et al., 1998), estimates of above- and below-ground biomass inputs were obtained from OP grown on mineral substrates (Henson & Dolmat, 2003; Lamade & Bouillet, 2005), and fluvial carbon fluxes were estimated using DOC and POC data from northern peatlands (Holden, 2005). Thus, although Murdiyarso et al. (2010) provide an appropriate analytical framework, the use of values obtained for ecosystems other than OP plantations on tropical peats (e.g., OP on mineral soils, northern peatlands) and from a range of spatially disparate sites in various stages of the OP production cycle limits their analysis. Attempting to balance inputs and outputs to and from the peat carbon pool using data from OP plantations of different age structure, above- and below-ground values from mature plantations on mineral soils (Henson & Dolmat, 2003; Lamade & Bouillet, 2005) and peat emissions from a relatively young five-year-old plantation (Melling et al., 2005b) for example, is clearly problematic and likely explains their low peat CO_2 emissions value.

In addition to the data limitations highlighted by Murdiyarso et al. (2010), a number of the other studies reviewed for this assessment also highlighted the limited empirical basis of their analyses (Reijnders & Huijbregts, 2008), as well as the general paucity of data specifically relating to GHG emissions from OP plantations on tropical peatland (Danielsen et al., 2009; Reijnders & Huijbregts, 2008). The values used to estimate peat emissions are based on a variety of sources, including IPCC (1997, 2006) defaults, closed chamber studies (from both OP and non-OP land uses) and subsidence monitoring. In most cases, an average value calculated from these various and disparate sources has been used as the basis for estimating CO_2 emissions without considering whether or not they are realistic or even correct.

In order to assess the empirical foundations of CO_2 emission estimates, we traced the values used to estimate peat surface CO_2 emissions back to the original publications. Our analysis was restricted to CO_2, as emissions of this gas are the most substantial component of the GHG budget of OP plantations and there is a paucity of empirical studies on non-CO_2 GHG emissions from OP plantations (and tropical peatlands more generally). Non-CO_2 GHG emissions are considered in more detail in section 5.4 below.

As an example of the propagation of emissions values through the literature, two studies reporting CO_2 emissions based on less than 50 measurements, using the closed chamber technique, at a few locations over one year (Murayama & Bakar, 1996; Melling et al., 2005b) are used to represent total long-term peat surface CO_2 emissions in the analysis of Murdiyarso et al. (2010) and have also now been extrapolated in estimating CO_2 emissions from the 800,000 ha of mature OP plantations on tropical peatland identified in a recent remote sensing analysis (Koh et al., 2011). As subsequent publications relating to OP biofuel feedstocks grown on tropical peatlands are published in higher-profile journals of increasing policy relevance, it is important that the uncertainties associated with these empirical measurements are understood and the reliability of the estimates assessed. The following section reviews the scientific publications used to estimate CO_2 emissions from drained peat under OP plantations.

5.1 Review of IPCC default emissions factors

A number of studies aiming to assess peat emissions for OP plantations on tropical peatland have used IPCC (1997, 2006) default emission factors (Fargione et al., 2008; Germer & Sauerborn, 2008; JRC, 2010; Wicke et al., 2008). JRC (2010) suggests that IPCC defaults have a limited empirical basis, reflecting the paucity of data available in 1996. For CO_2, the IPCC (1997, 2006) provide default emissions factors for agriculture on drained tropical organic soils (73.4 ± 66 Mg CO_2 ha^{-1} yr^{-1}) and managed forests on drained tropical organic soils (5 Mg CO_2 ha^{-1} yr^{-1}, with a range of 3 to 14 Mg CO_2 ha^{-1} yr^{-1}). In a number of studies, IPCC (1997, 2006) defaults were averaged with empirical estimates obtained from subsidence-based studies (JRC, 2010), gas fluxes from closed chamber measurements (Fargione et al., 2008) or a combination of the two (Germer & Sauerborn, 2008). It is clear, however, that some ambiguity exists in the application of IPCC (1997, 2006) defaults for estimating emissions from OP plantations on tropical peatlands.

Germer & Sauerborn (2008) combined the IPCC (1997) default for agriculture with the flux estimates of Melling et al. (2005b) and those based on subsidence estimates (Wösten et al., 1997) and peat bulk density values (Brown, Iverson, Prasad, & Liu, 1993). These authors assumed that emissions from OP plantations were one-quarter of the IPCC (2006) value for agriculture (i.e. 18.4 Mg CO_2 ha^{-1} yr^{-1}) on the grounds that OP cultivation requires less soil disturbance than settled farming; however, they also noted that this assumption likely underestimated CO_2 emissions as the deeper drainage depths required for OP cultivation were neglected. Similarly, JRC, (2010) averaged the IPCC (2006) default value for agriculture (73.4 Mg CO_2 ha^{-1} yr^{-1}) with a subsidence-based estimate provided by Couwenberg et al. (2010; see below) to estimate emissions of 57 Mg CO_2 ha^{-1} yr^{-1}. JRC (2010) also noted that the IPCC (2006) default was likely to underestimate emissions, as the deep drainage depths required for OP plantations were not considered. In the OP life-cycle analysis of Wicke et al. (2008), the IPCC (2006) defaults for agriculture and forest management on tropical organic soils were averaged to provide an estimate of 39.2 Mg CO_2 ha^{-1} yr^{-1}. In contrast to Germer & Sauerborn (2008) and JRC (2010), Wicke et al. (2008) justified their averaging on the basis that shallower drainage depths are required for OP plantations than for forest management on organic soils. In Fargione et al. (2008), the default emission value for agriculture was assumed to be representative of CO_2 emissions from drained peats under OP cultivation (averaged with closed chamber flux estimates to obtain a value of 55 Mg CO_2 ha^{-1} yr^{-1}).

5.2 Review of closed chamber flux measurements

A number of the studies reviewed above (section 5.1) included closed chamber measurement data in their estimates of CO_2 emissions from drained peats (Fargione et al., 2008; Germer & Sauerborn, 2008; Koh et al., 2011; Murdiyarso et al., 2010; Reijnders & Huijbregts, 2008). A summary of the closed chamber studies used by these studies to estimate peat CO_2 emissions is provided in Table 4. It should be noted that a number of these studies are the same as those used in combination with subsidence monitoring to develop a linear relationship between drainage depth and peat surface CO_2 emissions (Couwenberg et al., 2010; Delft Hydraulics, 2006; Hooijer et al. 2010) that is discussed in more detail in the following section. A recent study by Agus, Handayani, van Noordwijk, Idris, & Sabiham (2010) on CO_2 emissions from OP plantations, not cited in any of the publications reviewed in section 5.1, has also been included, as has a recent study demonstrating good practice in estimating peat CO_2 emissions using closed chambers (Jauhiainen, Hooijer, & Page, in review). With the exception of Reijnders & Huijbregts (2008), who assumed that an average of CO_2 emissions measured at abandoned agricultural and paddy fields and those at a secondary forest (Inubushi et al., 2003) could be assumed to be representative of OP plantations, all studies (reviewed in section 5.1) using closed chamber flux estimates cite those of Murayama & Bakar (1996) and Melling et al. (2005b) as the basis for CO_2 emissions from drained peat (Fargione et al. 2008; Koh et al., 2011; Murdiyarso et al., 2010).

Analysis of the closed chamber studies listed in Table 4 reveals a number of key issues relating to the use of these values as the basis for estimating peat CO_2 emissions from OP plantations in Southeast Asia. The analysis has been restricted to studies specifically reporting measurements made at plantations (Agus et al., 2010; Melling et al., 2005b; 2007a; Murayama & Bakar, 1996). In terms of regional coverage, the studies summarized in Table 4 collectively report emissions estimates obtained at only six individual OP plantations; the measurements of Melling et al. (2005b) and Melling et al. (2007a) are co-located. Spatial bias also exists, as measurements were obtained for OP plantations at just three locations across Southest Asia; measurements reported by Agus et al. (2010) were obtained at adjacent OP plantations.

At the site scale, most studies provide limited information on key site characteristics, such as: location on the peat dome (e.g., base, slope, or top), peat thickness and nutrient status, and land use history and drainage conditions. Descriptions of individual sampling locations and details of closed chamber method sample analysis are also generally lacking and, where replicate measurements are provided (Agus et al., 2010; Melling et al., 2005b), it is not apparent where individual measurements were obtained or how representative measurement locations were of peat surface conditions within individual plantations (e.g., microtopography, distance from trees, etc.).

From Table 4, it is apparent that the frequency of closed chamber measurements has been low and sampling typically biased towards certain parts of the diurnal (Melling et al., 2005b) and seasonal (Agus et al., 2010) cycle. While a few studies have attempted to address seasonal variation in CO_2 emissions (Melling et al., 2005a; 2007a), the rest have been conducted on a short-term basis (Agus et al., 2010; Murayama & Bakar, 1996). Diurnal or interannual variability in CO_2 exchange has not been systematically addressed. Murayama & Bakar (1996) provide two estimates of total peat surface CO_2 emissions made using a closed chamber technique at separate OP plantations in Malaysia. In the original publication, the authors provide two hourly flux values of 1.14 and 1.69 kg CO_2-C ha^{-1} hr^{-1}, made at single locations within plantations of unknown age, OP planting density, and land use history. Simple multiplication by time and conversion of C to CO_2 confirms this as the method used to obtain the values of 36.6 and 54.3 Mg CO_2 ha^{-1} yr^{-1} that were later cited by Alterra (2008), Fargione et al. (2008), and Murdiyarso et al. (2010). It is not clear at which point these values were extrapolated in this way, but it is evident that temporal upscaling from single point measurements is inappropriate, as any spatial or temporal (diurnal, seasonal, or interannual) variation in CO_2 emissions is ignored, and any methodological bias will be significantly amplified (Denmead, 2008). Whilst the total soil respiration estimates derived from Murayama & Bakar (1996) are clearly not robust estimates of annual CO_2 emissions, it is interesting to note that whilst Fargione et al. (2008) averaged both estimates with the 55 Mg CO_2 ha^{-1} yr^{-1} estimate of Melling et al. (2005b) and the IPCC (2006) default value for agriculture, Murdiyarso et al. (2010) averaged only the lower value derived from Murayama & Bakar (1996) with Melling et al. (2005b). It is unclear why only this lower-end value was selected, but this provides an additional explanation for the lower CO_2 emission estimate provided by Murdiyarso et al. (2010; Table 3).

The study by Melling et al. (2005b) has been widely cited in relation to OP plantation development on tropical peatland and has achieved notoriety due to the frequent misassumption that the closed chamber fluxes reported were representative of the net ecosystem CO_2 exchange (Alterra, 2008; Yew et al., 2010). Melling et al. (2005b) report total peat CO_2 emissions from peat swamp forest and OP and sago plantations in Sarawak, Malaysia. Coordinates provided in the paper and subsequent observations indicate that the forest site was affected by drainage during monitoring. For the OP plantation, Melling et al. (2005b) estimate undifferentiated CO_2 emissions of 55 ± 11 Mg CO_2 ha^{-1} yr^{-1} (the uncertainty range is based on the authors' suggestion of a 20% measurement error, and is most likely an underestimation). This estimate is based, however, on only three replicate measurements conducted once a month (i.e., 36 measurements in total), and the authors provide no information on how representative the sampling locations were of peat surface conditions within the plantation, or of the location of the plantation on the peat dome. Moreover, Melling et al. (2005b) report that the site experienced high year-round rainfall with no distinct dry period, suggesting measurements were done in an unrepresentative year and at a mean drainage depth of 0.6 m, which is at the lower end of values reported for OP plantations on tropical peats (Hooijer et al., 2010). A further questionable aspect of the study by Melling et al. (2005b) is the use of two-point linear regression in estimating fluxes (i.e. each flux was based on concentrations in two gas subsamples and the slope of the line between these two points). Clearly, if one or both gas samples were erroneous, this would significantly bias individual flux readings and the derived annual CO_2 budget from the small data set collected in the study.

Only three of the closed chamber studies summarized in Table 4 claim to report on differentiated (peat only) CO_2 emissions (Agus et al., 2010; Jauhiainen et al., in review; Melling et al., 2007a). In the case of Melling et al. (2007a), the estimate of 41 Mg CO_2 ha^{-1} yr^{-1} emission from peat only respiration at an OP plantation (at unknown drainage depth) was estimated using the trenching approach. According to Couwenberg et al. (2010), consideration of the errors associated with the closed chamber technique and trenching approach used by Melling et al. (2007a) suggests that emissions may be around 50 Mg CO_2 ha^{-1} yr^{-1}. In the study by Agus et al. (2010), it appears that the authors used only 30 cm deep collars inserted into the peat, and for quantification of root respiration, using another set of collars, they introduced three roots inside the collar through the hole in the collar wall and repacked the peat prior to the start of monitoring period. Measurements taken from closed chambers above the collars reported as representing peat respiration only (in the range of 18.2 to 24.3 Mg CO_2 ha^{-1} yr^{-1}) are in fact undifferentiated CO_2 emissions (with an unknown contribution from root respiration from below the shallow collar edge), and values reported as representing total soil respiration are actually total soil respiration from the disturbed peat profile plus contributions from artificially introduced pieces of root material of unknown dimensions (Agus et al., 2010). Although the chambers containing root material provided higher emission numbers in comparison to the non-perforated collars, the effect may have been caused by the disturbed peat and root material in the repacked collar, enhancing both autotrophic and heterotrophic emissions. The representativeness of introduced roots can also be questioned because the original root volumes in the peat around the OP stems or inside the collar are not reported. In addition, the long incubation time (35 minutes) in a relatively small chamber volume may provide another potential explanation. For these reasons, the estimates of Agus et al. (2010) should not be considered reliable estimates of total or differentiated peat CO_2 emissions.

Although its results were not obtained at an OP plantation, it is significant to highlight a recent study demonstrating good practice in estimating peat-only CO_2 emissions using closed chamber techniques (Jauhiainen et al., in review). At an *Acacia* plantation in Sumatra's Kampar Peninsula, Jauhiainen et al. (in review) estimated average annual daytime CO_2 emissions to be 93.9 Mg CO_2 ha^{-1} yr^{-1} at an average drainage depth of 0.8 m, or a minimum of 78 Mg CO_2 ha^{-1} yr^{-1} when corrected for diurnal variations in CO_2 emission. In contrast to the closed chamber studies reviewed previously, Jauhiainen et al. (in review) based their flux estimate on more than 2,300 measurements made at fortnightly to monthly intervals over two years, at 144 locations distributed over all peat dome zones. Measurements were made along transects of increasing peat thickness across a peat dome with a known land use history. The authors report measurements containing minimal contributions from root respiration, obtained using a combination of the trenching approach, and measurements made at varying distance from plantation trees (i.e., within and beyond the tree rooting zone) and before and after tree harvesting. Moreover, the CO_2 emission estimates from this study were found to be in good agreement with subsidence-based carbon loss measurements reported by Hooijer et al. (2011) for the same plantation landscape (discussed in more detail below). Furthermore, the Hooijer et al. (2011) study also reports subsidence and carbon loss values from OP plantations on deep peat in Jambi, Sumatra, which are nearly identical to those in *Acacia* plantations. This confirms our assessment that there are no fundamental differences in peat decomposition carbon emissions among *Acacia* and OP plantations on peatland. They share similar water management requirements, soil disturbance, and loss of natural canopy, although OP plantations are more heavily fertilized.

Table 4 (next page): Summary of available peat CO_2 emission data measured using the (dark) closed chamber technique for OP plantations on peatlands, and for peatlands where surface CO_2 emissions have been assumed to be representative of emissions from OP plantations in the studies reviewed in section 5.1. Total respiration values are the total emissions measured at the peat surface (root and peat respiration). Peat respiration values are CO_2 emissions assumed to be from the peat only. References marked with * were used by Hooijer et al. (2010) to develop a linear relationship between mean drainage depth and annual CO_2 emissions. Mean annual drainage depths are provided where values are indicated as being variable. Age of plantation, in years (where known) is shown. ND indicates data not available in original publications. All values originally presented in units of Mg C ha^{-1} yr^{-1} have been converted to Mg CO_2 ha^{-1} yr^{-1} using a factor of 3.67. A positive drainage depth denotes a water table above the peat surface. PSF = peat swamp forest.

REFERENCE	LAND USE (AGE)	LOCATION	TOTAL RESPIRATION Mg CO_2 ha^{-1} yr^{-1}	PEAT RESPIRATION Mg CO_2 ha^{-1} yr^{-1}	DRAINAGE DEPTH (m)	COMMENTS
Murayama & Bakar (1996)*	OP plantation (ND)	Central Selangor, Malaysia	36.6	ND	ND	No replicate measurements. Annual values cited in later publications have been obtained by multiplication of the single point estimates provided. No information on plantation age or drainage conditions provided.
	OP plantation (ND)	Western Johor, Malaysia	54.3	ND	ND	
Jauhiainen et al. (2001)*	Agriculture	Kalimantan, Indonesia	26	ND	0.5	Three plots on the site with eight monitoring location replicates at each (24 replicates). Data collected during daytime with portable infrared gas analyzer (IRGA). Data collection made 1-3x/week during 27.10.–20.11.1999; 30.3.–24.4.2000; 19.8.–10.9.2000; 9.3.–2.4.2001. Key characteristics provided; site developed ~20 yrs prior to monitoring, fire affected fallow peat without vegetation cover, no recent fertilization. Monitoring included water table depth and peat temperatures. Emission range and mean ± SE at 10 cm wide water table depth classes provided. Appended data used in subsequent publication.
Inubushi et al. (2003)	Abandoned upland field	Kalimantan, Indonesia	36.3 ± 4	ND	0.15 (Nov only)	Measurements made at monthly intervals between November 1999 and January 2001. Annual balance derived from monthly measurements. Three replicates made at each site per month. Data not obtained for OP plantations but assumed representative of OP by Reijnders & Huijbregts (2008). Drainage depth only reported for Nov. 2009.
	Abandoned paddy field		56.5 ± 15.8	ND	+0.021 (Nov only)	
	Secondary forest		44 ± 10.7	ND	0.1 (Nov only)	
Melling et al. (2005b)*	Sago plantation	Sarawak, Malaysia	40 ± 8	ND	0.27 (variable)	Three measurements made once a month, between 11:00 and 13:00, at each site over one year. Two-point regression used to estimate fluxes. Annual balance obtained by upscaling from hourly to monthly to annual values. Uncertainty range assumed from author's suggestion of 20% underestimation in flux measurements.
	Drainage-affected PSF		77 ± 15.4	ND	0.45 (variable)	
	OP plantation (five years)		55.1 ± 11	ND	0.6 (variable)	
Ali et al. (2006)*	Settled agriculture	Jambi, Indonesia	77	ND	0.78 (variable)	Two plots and five replicate measurements. Water table depth and temperature data collected. C, N, bulk density, and pH values provided. Measurements by portable IRGA. Measurements obtained throughout the day. Demonstrate a clear diurnal temperature response of surface CO_2 emission. Denotes mean CO_2 efflux correlation with water table depth.
Melling et al. (2007a)	OP plantation (five years)	Sarawak, Malaysia	56.5	40.9	Unknown	Trenching method used to differentiate peat and root respiration. Five-year-old oil palms on peat 5.55 m thick. C (44.7%), N (2%) and loss of ignition (99%). Measured at monthly intervals for a year. Three permanent replicates made at field. CO2 determined in the laboratory by IRGA (no other information on analytical procedures). Provides single values for the two emission types.
Agus et al. (2010)	OP plantation (one year)	Nanggroe Aceh Darussalam Province, Indonesia	40.9 ± 18	24.3 ± 9.7		Measurements made during October and November only, and between 07:00 and 10:00. Annual balance (and standard deviations) extrapolated from the mean of all measurements. Inappropriate closed chamber methodology. OP roots added to chamber covered area rather than being excluded.
	OP plantation (five years)		27.3 ± 5.6	18.2 ± 11.1		
	OP plantation (10 years)		32.9 ± 20.7	19.3 ± 16.6		
Jauhiainen et al. (in review)	Acacia plantation (eight to 10 years)	Kampar Peninsula, Riau Province, Sumatra	102.5 ± 27.8	78 to 93.9 ± 17.2	0.8	More than 23,00 measurements obtained at max. monthly intervals between April 2007 and April 2009. Measurements made along eight transects across a peat dome of known land use history. Root respiration mostly excluded using a combination of trenching, and measurements at variable distances from plantation trees. Estimates of root respiration contribution to total peat emission at measurement locations provided.

5.3 Review of subsidence-based estimates

As with the closed chamber flux measurements discussed in section 5.2, most subsidence-based studies provide insufficient detail of site conditions; the numbers of replicates are also typically very low. From Table 2, it is evident that subsidence-based estimates of peat carbon loss have not been widely used in previous studies aiming to assess emissions from OP plantations on tropical peats[6]. Of these studies, only Germer & Sauerborn (2008) and the IFPRI-MIRAGE model (Al-Riffai et al., 2010) include estimates based on subsidence monitoring in their analyses. Germer & Sauerborn (2008) estimated peat CO_2 emissions on the basis of calculations provided by Wösten et al. (1997). Assuming a (projected) average annual subsidence rate of 2 cm yr^{-1}, peat bulk density of 0.1 g cm^{-1}, a peat carbon content of 60%, and 60% of subsidence ascribed to decomposition, Germer & Sauerborn (2008) estimate peat emissions to be 26.4 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}. They also use the bulk density value of 0.15 g cm^{-1} and peat carbon content of 45% provided by Brown et al. (1993) to estimate emissions of 29.7 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}. These values were averaged with the IPCC (1997) and closed chamber flux estimates (described in sections above) to arrive at peat emissions of 31.4 ± 14.1 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}.

The Wösten et al. (1997) study used by Germer & Sauerborn (2008) provides a linear relationship for estimating CO_2 emissions from drained tropical peatlands. According to later publications (Couwenberg et al. 2010; Delft Hydraulics, 2006; Hooijer et al. 2010; Wösten et al., 2008), the Wösten et al. (1997), relationship estimates emissions of between 13 and 39 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for each additional 10 cm drainage depth depending on the peat bulk density value used in the model (0.05 to 0.15 g cm^{-3}, respectively). It appears that Germer & Sauerborn (2008) misinterpreted Wösten et al. (1997) by using a projected value for subsidence (JRC, 2010), rather than the observed subsidence rate of 4.5 cm yr^{-1} on which the relationship cited by later publications is based (Couwenberg et al. 2010; Delft Hydraulics, 2006; Hooijer et al. 2010; Wösten et al., 2008). Clearly, as OP cultivation requires repeat drainage, the use of the projected value is inappropriate (JRC, 2010). The Wösten et al. (1997) relationship assumes an oxidative component of 60% and a peat bulk carbon content (bulk density of 0.1 multiplied by 60% carbon content) of 0.06 g C cm^{-3}. Applying the 13 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} estimate for each additional 0.1 m drainage (Wösten et al., 2008) to typical OP plantation drainage depths of 0.6, 0.85 and 1 m, the linear relationship provided by Wösten et al. (1997) estimates peat emissions of 78, 110.5, and 130 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}, respectively.

In the JRC iLUC modeling comparison (JRC, 2010), peat CO_2 emissions were estimated using a linear relationship between drainage depth and the average emissions of Couwenberg et al. (2010) and the IPCC (2006)

6 The studies by Gibbs et al. (2008) and Danielsen et al. (2009) cite Hooijer et al. (2006) as the basis for their peat carbon emissions. As these publications do not, however, provide annual $CO_{2\text{-eq}}$ emission values, they have been excluded from the present discussion.

default for agriculture (JRC, 2010). This relationship, derived on the basis of a literature review, predicts that peat carbon emissions increase by 9 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for every 0.1 m of additional drainage depth for a range of land use types. Parameters used in the model are a peat carbon density of 0.068g C cm^{-3} and 40% decomposition. Couwenberg et al. (2010) suggest that the linear function is valid up to and including drainage depths of 0.5 m, after which subsidence rates (and carbon losses) level off. From Figure 2 in Couwenberg et al. (2010), however, it appears that few data were available for drainage depths beyond 0.5 m. Moreover, more comprehensive field measurements indicate subsidence may not level off until drainage depths of up to 1 m (Hooijer et al., 2008; Hooijer et al., 2011). At 0.5 m drainage depth, 0.1 m lower than the minimum (0.6 m) drainage depth typically required for OP (JRC, 2010; Alterra, 2008), the Couwenberg et al. (2010) relationship predicts peat emissions of 45 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}. Assuming subsidence is linear up to drainage depths approaching 1 m, emissions at typical plantation drainage depths of 0.6 and 0.85 m would be 54 to 77 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}, respectively, and 90 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} at drainage of 1 m.

The Couwenberg et al. (2010) model is based on the minimum peat decomposition value (40%) reported for tropical peatlands (Murdiyarso et al., 2010; Murayama & Bakar, 1996; Wösten et al., 1997) and a peat bulk carbon density of 0.068 g C cm^{-3}. If instead decomposition is assumed to be 60% of subsidence, and assuming subsidence does level off at drainage depths beyond 0.5 m, recalculation using the Couwenberg et al. (2010) model predicts peat emissions of 67 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}. If the subsidence rate remains a linear function of drainage up to 1 m, then emissions of 81, 115 and 135 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} are predicted at 0.6, 0.85, and 1 m drainage depths, respectively. Additionally, the peat carbon density value of 0.068 g C cm^{-3} used by Couwenberg et al. (2010) does not account for the increased peat bulk density of surface peat in OP plantations reported in some studies (JRC, 2010). In a recent study, Ywih, Ahmed, Majid, & Jalloh (2010) indicate that the carbon density in OP plantations in Sarawak, Malaysia was relatively constant at approximately 0.134 g C cm^{-3} to depths of 0.5 m up to five years after land use conversion. Using this value in the Couwenberg et al. (2010) model, at a drainage depth of 0.5 m and using conservative decomposition values of 40 to 60%, peat emissions are then estimated to be 89 to 133 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}, respectively. Clearly, these values would be significantly higher if subsidence continued beyond 0.5 m; however, calculations are restricted to 0.5 m because the values reported by Ywih et al. (2010) are for this depth only.

An additional relationship between drainage depth and peat carbon loss is provided by Delft Hydraulics (2006) and Hooijer et al. (2010), who combined the subsidence-based estimate of Wösten et al. (1997) and Wösten & Ritzema (2001) with published estimates of peat CO_2 emissions from a variety of drained peatlands. From this, they derived a linear relationship predicting peat emissions of 9.1 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for each additional

10 cm drainage depth. On the basis of this relation, Delft Hydraulics (2006) and Hooijer et al. (2010) estimate emissions of 73, 86 and 100 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} at OP plantation drainage depths of 0.8, 0.95 and 1.1 m, respectively. Appling this relationship to the typical OP plantation drainage depths of 0.6, 0.85 and 1 m as above yields emissions estimates of 54.6 to 77.35 and 91 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}, respectively. Although of similar magnitude to the relationship provided by Couwenberg et al. (2010), Delft Hydraulics (2006) and Hooijer et al. (2010) suggested that this relationship required further refinement as more field data, particularly under different land uses and at different times since the start of drainage, became available. They also stated that while the linear relationship was considered the best estimate currently available for determining carbon loss at water table depths between 0.5 and 1 m, it could prove to be curved, although this would make little difference to estimates of peat carbon loss at typical plantation water tables around one metre below the surface. The Couwenberg et al. (2010) and earlier Delft Hydraulics (2006) and Hooijer et al. (2010) regressions were both forced through zero, although the more recent study by Hooijer et al. (2011) suggests that in deforested peat, zero subsidence may not be reached even under water saturated conditions. The study of gaseous CO_2 emissions from peat by Jauhiainen et al. (in review) also supports this view.

This recent study by Hooijer et al. (2011) provides comprehensive subsidence-based estimates of peat carbon emissions from several OP and *Acacia* plantations in Sumatra. Based on over 200 subsidence measurements (more than were previously available for all peatlands in Southeast Asia combined), taken at various locations across individual plantations and peat domes combined with bulk density profiles, Hooijer et al. (2011) estimate that decomposition contributes at least 92% to monitored long-term subsidence rates, much higher than previous estimates. They report initial subsidence rates of 1.42 m during the first 5 years, followed by secondary subsidence rates averaging 5 to 5.4 cm yr^{-1} at mean drainage depths of 0.7 to 0.73 m for *Acacia* and OP plantations, respectively. These observed subsidence rates suggest that subsidence continues to increase beyond the 0.5 m drainage depth suggested by Couwenberg et al. (2010). For OP, however, these authors were unable to identify a relationship between subsidence rate and drainage depth, possibly because decomposition of drained peat in OP plantations may be operating at maximum rates regardless of water levels owing to high rates of nitrogen fertilizer application (*ca*. 0.5 Mg N ha^{-1} yr^{-1}). Using data obtained from all (*Acacia* and OP) plantations, they estimated average peat carbon, the highest emissions, of 178 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}, occurring during the first five years following drainage, followed by lower emissions of 70 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} in subsequent years. Accounting for initially high rates, Hooijer et al. (2011) estimate average plantation emissions (for *Acacia* and OP plantations combined) of 100 and 86 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} when annualized over 25 and 50 years periods, respectively. The average peat carbon emission estimate of at least 70 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}

for plantations that have been drained for more than five years is supported by closed chamber flux estimates of 78 to 94 Mg CO_2 ha^{-1} yr^{-1} obtained for *Acacia* plantation at the same area (Jauhiainen et al., in review). Although not specific to the OP plantations monitored for subsidence, the findings from closed chamber measurements support the general validity and quality of the subsidence monitoring results of Hooijer et al. (2011). On the basis of the same analysis, Hooijer et al. (2011) also estimate additional carbon losses of 33 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} from drainage-affected forests in the areas adjacent to the plantations monitored, over a zone of 2 km around perimeter canals.

The estimates of peat carbon emissions based on the review of subsidence monitoring presented above are summarised in Table 5, with annualized values in Table 3. The annualized values take into account the higher emissions during the first few years following drainage. Peat carbon emissions have been estimated for a range of plantation drainage depths, 0.5 m [to account for the possibility that subsidence levels off at 0.5 m (Couwenberg, 2011)], optimal plantation drainage depths of 0.6 to 0.85 m, and drainage of 1 m (on the assumption that subsidence rates reach a maximum at this drainage depth and level off thereafter). For comparison, emissions estimated for the mean drainage depth of 0.7 m reported in the study of Hooijer et al. (2011) have also been included. Assuming that subsidence rates level off at 0.5 m (Couwenberg et al., 2010), and accounting for possible increased peat bulk density under OP, estimates of peat carbon emissions are in the range of 45 to 133 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}. The upper value is close to the maximum emission value of 135 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} predicted at 1 m using a value of 60% for the peat decomposition contribution to subsidence in the Couwenberg et al. (2010) model. Using this same model and assuming that subsidence rates continue up to 1 m, but excluding potential increases in peat bulk density following conversion to OP because the model is only valid to a depth of 0.5 m, more typical plantation drainage depths of between 0.6 and 0.85 m result in peat carbon emissions in the range of 54 to 115 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}. The recent study of Hooijer et al. (2011) falls within this range. It is important to note that that study is the only one to provide data that enable calculation of annualised values for peat carbon emissions over various time scales, thereby accounting for higher rates of emission in the years immediately following drainage. Earlier studies had assumed constant emissions since plantation development.

Table 5 (next page): Summary of peat carbon emissions estimated for various drainage depths on the basis of subsidence monitoring. See the main text for descriptions of the calculations and values used.

REFERENCE	DESCRIPTION	CO_2 EMISSION (Mg CO_2 ha^{-1} yr^{-1}) AT DIFFERENT PLANTATION DRAINAGE DEPTHS (cm)				
		50	60	70	85	100
Wösten et al. (1997)	Relationship predicts emissions of 13 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for each additional 0.1 m drainage depth. Based on subsidence rate of 0.45 m yr^{-1}, 60% decomposition, bulk C density of 0.06 g C m^{-3} .	65	78	91	110.5	130
Delft Hydraulics (2006) and Hooijer et al. (2010)	Relationship predicts emissions of 0.91 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for each additional 0.1 m drainage depth. Model is based on the subsidence model of Wösten et al. (1997) combined with closed chamber measurements.	45.5	54.6	64	77.4	91
Couwenberg et al. (2010)	Original model. Predicts emissions of 0.9 Mg $CO_{2\text{-eq}}$ ha^{-1} from each additional 0,1 m drainage depth, assuming 40% decomposition and a bulk carbon density of 0.068 g C cm^{-3}.	45	54	63	77	90
	Decomposition contributes 60% of subsidence, bulk carbon density of 0.068 g C cm^{-3}.	67	81	94	115	135
	Decomposition contributes 40%, bulk carbon density of 0.138 g C cm^{-3} in upper 0.5 m of peat profile; from Ywih et al. (2010), values only calculated for drainage of 0.5 m.	89	—	—	—	—
	Decomposition contributes 60%, bulk carbon density of 0.138 g C cm^{-3} in upper 0.5 m of peat profile; from Ywih et al.(2010), values only calculated for drainage of 0.5 m.	133	—	—	—	—
Hooijer et al. (2011)*	Empirical estimate of CO_2 loss due to drainage. Decomposition estimated to be 92% of subsidence. Carbon loss over first five years after plantation drainage is higher (178 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}). Values provided are emissions annualized over 25 and 50 years (see also Table 3).	—	—	86 - 100	—	—
Summary	**Mean (standard deviation)**	**74 (33)**	**67 (15)**	**81 (17)**	**95 (21)**	**111 (24)**
	Median	**66**	**67**	**86**	**94**	**111**
	Minimum	**45**	**54**	**63**	**77**	**90**
	Maximum	**133**	**81**	**100**	**115**	**135**

Table 6: Annualized values for peat carbon emissions from plantations over various time scales, accounting for higher rates of emission in the years immediately following drainage. Average water table depth 0.7 m; values derived from Hooijer et al. (2011).

NUMBER OF YEARS	CO_2 EMISSION ($Mg\ CO_{2\text{-eq}}\ ha^{-1}\ yr^{-1}$)
5	178
10	121
20	106
25	100
30	95
40	90
50	86

5.4 Review of non-CO_2 greenhouse gases (CH_4 & N_2O)

Similar to studies on peat CO_2 emissions, there have been a limited number of studies reporting on fluxes of CH_4 and N_2O from OP plantations on tropical peats (Couwenberg et al., 2010). All have used the closed chamber methodology to estimate fluxes, and no eddy covariance measurements of CH_4 or N_2O have yet been reported for OP plantations on tropical peat. The following sections review the values used to estimate peat CH_4 and N_2O fluxes in the studies reviewed in section 5.1 and the limited number of available empirical studies made in OP plantations. Although there are other estimates of non-CO_2 GHG fluxes in peat under other land use types (e.g. agricultural sites on peat), the focus here is on OP plantations, and these other studies are not considered below. See Couwenberg et al. (2010) for a comprehensive review of soil non-CO_2 GHG fluxes from other land use categories.

5.4.1 METHANE (CH_4)

Few of the studies reviewed in section 5.1 consider peat CH_4 fluxes in their assessments of GHG emissions from OP plantations. Only Germer & Sauerborn (2008) consider the consumption of CH_4 in the drained peat profile in their analysis, basing their estimate on the study of Melling et al. (2005a). The latter provides one of the few estimates of CH_4 flux in an OP plantation on tropical peat. Annual peat CH_4 fluxes were estimated to be -15.14 Mg C m^{-2} yr^{-1} and, similarly to their estimates of annual CO_2 emissions at the same site (Melling et al., 2005b), the annual peat CH_4 balance was obtained using the closed chamber technique. Measurements were made over one year at the peat surface, using three replicate monthly measurements made between 11:00 and 13:00, and using a linear regression between two points (Melling et al., 2005a). As with the CO_2 emissions estimates of Melling et al. (2005b), it is not clear where measurements were made within the OP plantation, i.e. location on the peat dome, or indeed how representative the site is of OP plantations in Southeast Asia more generally.

Despite the obvious limitations in the experimental design of Melling et al. (2005a), it is generally accepted that peat CH_4 fluxes in drained tropical peatlands are insignificant relative to losses of CO_2, both in terms of the mass of carbon lost and overall possible climatic impact (Couwenberg et al., 2010; Murdiyarso et al., 2010). Moreover, modest CH_4 consumption is often observed at drainage depths below 0.2 m (e.g. Jauhiainen et al., 2008, 2005). It should be noted however, that water surface CH_4 emissions from the network of ditches and canals required to drain OP plantations may be significant, and such hot spots of CH_4 emission have been observed in temperate and boreal peatlands (e.g. Hendricks et al., 2007; Teh et al., 2011). As CH_4 emissions are likely to scale linearly with the density of the drainage network, this potentially important source of CH_4 remains to be quantified and should not be considered negligible (Alterra, 2008; Jauhiainen et al., 2010).

5.4.2 NITROUS OXIDE (N_2O)

Only two of the studies reviewed in section 5.1 considered soil N_2O fluxes in their analysis of GHG emissions from OP plantations on peat (Germer & Sauerborn, 2008; Wicke et al., 2008). In Germer & Sauerborn (2008), peat emissions were estimated to be 4.1 ± 5.5 kg N_2O-N ha^{-1} yr^{-1} (1.2 ± 1.6 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}). This estimate was obtained by averaging values obtained for a histosol planted with corn in Florida (Terry, Tate & Duxbury, 1981) with N_2O emissions measured at a cassava plantation on peat and a secondary peat swamp forest in Indonesia (Hadi et al., 2000). Interestingly, Germer & Sauerborn (2008) attempted to base their estimate on N_2O emission from the drained peat alone, excluding N_2O emissions resulting from nitrogen fertilizer applications. It is not clear why this was attempted, as emissions resulting from fertilization should be attributed to the anthropogenic modification of the peatland GHG balance. In the analysis of Wicke et al. (2008), peat N_2O emissions were estimated to be 8 kg N_2O-N ha^{-1} yr^{-1} (3.7 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}) using the IPCC (2006) default for agroforestry on tropical peats. According to Wetlands International (2009a), this default value should be revised to 3.4 (range: -0.5 to 13.4) kg N_2O-N ha^{-1} yr^{-1} on the basis of currently available data, or 1.6 (range: -0.23 to 6.27) Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}.

In the only study to report annual peat N_2O balance from an OP plantation on tropical peatland, Melling et al. (2007) report annual emissions of 1.2 kg N_2O ha^{-1} yr^{-1} (0.56 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}) at an OP plantation fertilized by two annual applications of 51.5 kg N ha^{-1} in the form of urea. Based on the (undifferentiated) CO_2 measurements of Melling et al. (2005b), this equates to about one percent of peat surface CO_2 emissions and is in the lower range of the revised values for agroforestry on tropical peatland (Wetlands International, 2009a). As with previous studies, Melling et al. (2005a, b) estimated annual balances on the basis of three replicate measurements made at monthly intervals over one year, with all measurements made between 11:00 and 13:00. As such, considering the spatially and temporally highly variable nature of N_2O flux dynamics in response to environmental conditions, the results of this study cannot be considered a reliable estimate of annual peat N_2O balance. Moreover, it is unlikely that emissions resulting from the two fertilization applications were adequately captured by monthly measurements, since there is likely a rapid post-fertilization emission pulse (Jauhiainen et al., in review).

5.5 Emissions reductions with good land management practice

In addition to the general paucity of data on peat surface GHG emissions from OP plantations on tropical peatlands, limited research has focused on minimizing emissions through good agricultural practice. To date, the few publications that have touched on this topic have suggested optimized hydrological (i.e., drainage), management as a means of reducing emissions (Delft Hydraulics 2006; Hooijer et al. 2010) or establishment of hydrological buffer zones to minimise the impacts of drainage on forested peatland surrounding plantations (Hooijer et al., 2011). At the time of writing, and given the general lack of data on emissions from OP plantations in general, it is unsurprising that there appears to be no published information on GHG emissions from OP plantations that have adopted such mitigation strategies. Although Hooijer et al. (2011) suggest that optimizing plantation drainage depths at the minimum level required for OP cultivation (0.6 m) could potentially achieve peat CO_2 emissions reductions of up to 20%, they also stress that this would only serve to delay the inevitable peat carbon loss that is associated with drainage (Hooijer et al., 2011). The effects of optimized water table management on CH_4 and N_2O dynamics remain unquantified based on data from OP plantations. Effective use of hydrological buffer zones could also be used to minimize peat CO_2 emissions from forested areas surrounding plantations [estimated as 33 Mg CO_2 ha^{-1} yr^{-1} for distances of up to 2 km from the forest-plantation boundary by Hooijer et al. (2011)].

The ecological restoration of OP plantations to peat swamp forest by revegetation and rewetting the drained peat may offer a further option for GHG emission reductions (Hooijer et al., 2011; Koh et al., 2011). Studies on the impacts of restoration on tropical peatland GHG balances, however, remain in their infancy (e.g. Jauhainen et al., 2008a; Page et al., 2009) and have not yet been conducted for OP plantations. Moreover, given the social and economic drivers of OP plantation expansion in SE Asia (Renewable Fuels

Agency, 2010), coupled with the current profitability of OP production over initiatives such as REDD+ (Butler, Koh & Ghazoul, 2009), it is unlikely that the restoration of existing OP plantations will become widespread in the near term.

The conclusion from the limited information currently available is that while optimized drainage management and hydrological buffer zones may be effective in reducing GHG emissions in the near-term, the only effective means of mitigating emissions from OP plantations on tropical peat is to minimize the area of tropical peatland converted for production (Hooijer et al., 2011; Paoli et al., 2011). In terms of existing OP plantations on peat, it must therefore be accepted that high rates of GHG emission from peatland drainage represent the inevitable cost of OP cultivation. According to Fargione et al. (2008), Germer & Sauerborn (2008), and Wicke et al. (2008), OP biofuel production could provide reduced emissions, but only if OP feedstocks are produced on degraded (non-peat) lands or anthropogenic grasslands. Given some of the current data limitations identified in this review, and wider concerns over indigenous land rights and biological diversity (Danielsen et al., 2009; Renewable Fuels Agency, 2010), caution should be exercised before any particular existing land use is preferentially targeted for OP expansion, although using OP to rehabilitate anthropogenic *Imperata* grasslands seems to have significant potential (Ecofys, 2009). Moreover, the term “degraded land” should be carefully defined, so as to exclude any degraded peatland areas as well as areas of logged-over forest, which may still retain high value for biodiversity support and provision of other ecosystem services.

6 CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

This report has aimed to assess likely rates of GHG emission from tropical peatlands converted to OP plantations and to provide a best estimate and uncertainty profile. The review has focused primarily on CO_2 emissions as this gas is widely considered to be the most significant part of the overall carbon and GHG balance from drained tropical peats (Couwenberg et al., 2010; Jauhiainen et al., 2011; Murdiyarso et al., 2010). Previous assessments have based their estimates of CO_2 emission on a variety of sources, including IPCC (1996, 2006) defaults, closed chamber flux estimates and subsidence monitoring. In terms of IPCC defaults, the results from available closed chamber and subsidence studies indicate that the use of these values (range of 3 to 14 Mg CO_2 ha^{-1} yr^{-1}) for managed forests (agroforestry) on tropical peats is inappropriately low (e.g. Wicke et al., 2008), as are estimates based on one-quarter of the value for agriculture on drained peats (e.g. Germer & Sauerborn, 2008). Previous analyses incorporating these values into estimates of peat CO_2 emissions have significantly underestimated actual rates from OP plantations. Furthermore, although the IPCC (2006) default for agriculture on tropical peat is within the range predicted on the basis of subsidence monitoring (Table 2; Couwenberg et al., 2010; Delft Hydraulics, 2006; Hooijer et al., 2010; Wösten et al., 1997), these values have limited empirical foundation (JRC, 2010), having been developed for Tier 1 GHG assessments (IPCC, 2006), and their continued application in estimating CO_2 emissions from OP plantations on tropical peatland should be discouraged in favour of more appropriate, empirically based values.

There have been a number of studies using closed chamber methods to assess peat CO_2 emissions from OP plantations, but these have been limited in a number of aspects (section 5.2), and it remains uncertain how representative these values are for OP plantations across the Southeast Asian region. The 56 Mg CO_2 ha^{-1} yr^{-1} estimate of Melling et al. (2005b), for example, was obtained at a site with a mean drainage depth at the lower end required for OP cultivation (0.6 m), and in an area (coastal Sarawak) that was experiencing particularly high annual rainfall.

Several recent studies have applied subsidence data in order to provide a robust means of estimating carbon losses arising from the decomposition of drained peats (Couwenberg et al., 2010; Hooijer et al., 2011; JRC, 2010; Wetlands International, 2009a). This approach has several advantages in that it is relatively cheap and methodologically less complex than the closed chamber method; it can, however, require a relatively long time series of data, and care must be taken to obtain accurate measures of peat bulk density and carbon concentration (section 5.2). On the basis of this review, mean (± SD) estimates based on subsidence monitoring are in the range of 67 ± 15 to 95 ± 21 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for water table depths in the range of 60 – 85 cm, with (non-annualised) minimum and maximum values of 54 to 115 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}. Using this approach, peat carbon emissions annualized over 25 and 50 years are estimated to be 100 and 86 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}, respectively, taking into account very high emissions during the first 5 years following plantation drainage. It would appear, therefore, that the majority of previous studies aiming to assess GHG emissions from OP production systems on tropical peatlands have at best based their analyses on values below or towards the lower end of this range, and at worst have significantly underestimated CO_2 emissions from drained peats. In terms of biofuel production, therefore, it is likely that the true magnitude of the biofuel carbon debt for OP feedstocks produced on tropical peatlands is more substantial than has been previously assumed (Fig. 11; Table 2).

In terms of non-CO_2 GHG emissions (CH_4 and N_2O), there have been a very limited number of studies aiming to quantify annual emissions from peat. All support the general consensus that peat surface flux (emission or consumption) of CH_4 is typically low following land use conversion to OP plantation, but no studies provide data for emissions from the plantation drainage network, which has been noted as a point source for higher emissions (Jauhiainen, unpublished data). It is therefore unlikely that the true magnitude of or spatial and temporal dynamics of CH_4 emissions in OP plantations have been adequately captured by the limited number and frequency of measurements available, and more studies are required. Similar to peat CH_4 fluxes, the likely flux balance of N_2O at OP plantations on tropical peatland remains uncertain. To date, there appears to have been only one study aiming to quantify emissions of N_2O from OP (Melling et al., 2007). As with studies of CH_4 flux, the frequency and limited spatial context of these measurements are unlikely to have adequately captured the true magnitude and dynamics of peat N_2O fluxes at plantations, particularly in the periods following fertilizer applications. The estimate provided by Melling et al. (2007) indicates that N_2O emissions are a small component (about one percent) of the overall OP plantation GHG balance, but more measurements are required. Additionally, whilst N_2O emissions were not considered by the majority of previous analyses of GHG emissions from OP production systems (section 5.4), given the potency of this GHG, and in the absence of more comprehensive datasets, it is recommended that future assessments incorporate the range of revised peat N_2O flux balances (-0.5 to 13.4 kg N_2O-N ha^{-1} yr^{-1}) provided in the review by Wetlands International (2009a) as a means of better constraining uncertainties. As with more effective assessments of CO_2 emissions, accurate measurements of both CH_4 and N_2O fluxes would benefit substantially from the continuous and long-term monitoring capabilities provided by the eddy covariance technique, or the use of other more advanced measurement techniques (Denmead, 2008; Wang et al., 2011).

6.2 Recommendations

From the review above, we suggest that the annualization-dependent results of Hooijer et al. (2011) represent the most robust currently available empirical estimate of peat CO_2 emissions from OP plantations, the validity of the estimate being supported by the conclusions from the closed chamber measurement study of Jauhiainen et al. (in review). For a 50-year annualization of peat carbon emissions, Hooijer et al. (in press) give a value of **86 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}**. For a 25 year annualization, the value would be **100 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}**. In the context of modeling indirect land use change, modelers might be interested in using a 20- or 30-year annualization to match the land use change amortization periods in European and Californian biofuel legislation – these would give **106 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}** and **95 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}** respectively. These estimates explicitly account for higher peat carbon emissions observed in the early stages of drainage. In terms of an uncertainty range, we suggest that the likely peat carbon loss rate should be represented by the minimum and maximum values of **54** to **115 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}** for drainage depths of 0.6 to 0.85 m (Table 5), respectively. The minimum value of 54 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} is independently suggested by the linear relationships of Couwenberg et al. (2010), Delft Hydraulics (2006) and Hooijer et al. (2010). The upper value additionally accounts for potentially higher CO_2 emissions due to the reported higher peat carbon density in OP plantations. This upper value is also close to the median value of 111 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} predicted for drainage depths of 1.1 m. It should be noted that these values do not explicitly consider regional or local geographical variations (e.g., climate, location on the peat dome) or other factors (e.g., fertilization, land use history) promoting peat GHG emissions. The adoption of the best estimate and full uncertainty range suggested here will, however, lead to reduced uncertainty in future assessments conducted at the regional scale.

Given the increasing global demand for palm oil and existing land constraints in many parts of Southeast Asia, it appears likely that tropical peatlands will continue to be converted for OP production in the near future. The findings of this review suggest that GHG emissions from this rapidly expanding land use have been underestimated by previous assessments. The likely underestimation of emissions from peat in previous assessments has implications for the results of the modeling of the land use impacts of biofuel policies, and hence potentially for the policies themselves. The underestimation or non-inclusion of peat emissions from oil palm expansion in most previous modeling of the iLUC impacts of biofuels was noted by JRC (2010). Based on this review, the value of 57 Mg CO_2 ha^{-1} yr^{-1} proposed by JRC (2010) is also an underestimate (although we note that these authors also propose an upwards revised value of 112 Mg CO_2 ha^{-1} yr^{-1}, which may be an overestimate). This underestimation of peat GHG emissions in the iLUC modelling literature may have contributed significantly to an underaccounting of the indirect land use change GHG emissions of biodiesel, and in particular of biodiesel made from palm oil. For instance, Al-Riffai et al. (2010) used two emission values — 5 and 40 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1}, based on IPCC

(2006), and Wetlands International (2009a), averaged to 22.5 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} — to find that peat emissions contributed around 4 g $CO_{2\text{-eq}}$ MJ^{-1} to the carbon intensity of palm biodiesel, and perhaps under 1 g $CO_{2\text{-eq}}$ MJ^{-1} to the carbon intensity of other biodiesel. With the central value suggested here, those values would have been more like 19 and 5 g $CO_{2\text{-eq}}$ MJ^{-1}, respectively. JRC (2010) noted that the estimate of 18% of OP expansion occurring at the expense of peat had also been set too low by Al-Riffai et al. (2010). In that case, correcting up to 33% as suggested by JRC (2010) would create a compound effect and further increase the reported peat contribution to the biodiesel carbon intensities to 35 and 9 g $CO_{2\text{-eq}}$ MJ^{-1} for palm oil biodiesel and other biodiesel, an intensity increase of 31 and 8 g $CO_{2\text{-eq}}$ MJ^{-1}, respectively. To place this in context, an increase in carbon intensity of 31 g $CO_{2\text{-eq}}$ MJ^{-1} would subtract 37% from the reportable carbon savings of palm oil biodiesel used in the European Union.

More generally, it is already well established that the emissions from the destruction of Southeast Asian peat swamp forests are very substantial, and this review suggests that global emissions may be significantly higher than previously estimated. Wetlands International (2009b), based on Wetlands International (2009a), estimated global emissions from peatland (including fires) to be potentially in excess of 1.3 Gt yr^{-1}, with about 550 Mt yr^{-1} of that coming from degrading peatland in Indonesia and Malaysia. Given that Wetlands International (2009b) assumed a value of 40 Mg CO_2 ha^{-1} yr^{-1} for agroforestry, cropland, and grassland, it is suggested that the absolute level of peatland emissions from these countries and other tropical countries may have been underestimated substantially (noting that we have not explicitly reviewed emissions from other forest/agroforestry, grassland, and agriculture on degrading peat, but that these values may well have been subject to similar underestimation to the value for OP). That could put global emissions from degrading tropical peat in excess of 2 Gt yr^{-1}, although a more comprehensive review of the literature for degrading peat with non-OP uses would be necessary before a revised figure could be firmly suggested. ICCT (in press), in a forthcoming study, finds that between 1990 and 2010, the area of OP on peat in Malaysia and Indonesia combined increased by about 1.9 Mha. Based on the 25-year annualization, the annual emissions from this area would currently be about 200 Mt yr^{-1}, perhaps one-tenth of total global peat emissions, and two-fifths of the total for Indonesia and Malaysia. ICCT (in press) suggests that OP is likely to continue to expand rapidly onto peat in the next decade, in which case this rate of emissions could double by 2020.

All of this serves to emphasise that governments concerned with preventing climate change should act as a matter of urgency to protect global peatlands, primarily by restricting agricultural expansion on peatland and also by considering opportunities to rewet and restore already degrading peatlands. Effective protection would have the side benefit of significantly reducing the iLUC emissions intensity of biodiesel, potentially making biodiesel support mandates more viable as GHG mitigation policies.

REFERENCES

Agus, F., Handayani, E., van Noordwijk, M., Idris, K., & Sabiham, S. (2010). Root respiration interferes with peat CO_2 emission measurement. In *19th World Congress of Soil Science, Soil Solutions for a Changing World*. Brisbane, Australia.

Al-Riffai, P., Dimaranan, B. & Laborde, D. (2010). Global trade and environmental impact of the EU biofuels mandate [online]. Final Report for EC – DG Trade. Retrieved from http://trade.ec.europa.eu/doclib/docs/2010/march/tradoc_145954.pdf.

Ali M., Taylor D. & Inubushi K. (2006). Effects of environmental variations on CO_2 flux from a tropical peatland in eastern Sumatra. *Wetlands*, 26(2), 612-618.

Andriesse, J. P. (Ed.) (1988). *Nature and Management of Tropical Peat Soils*. Italy, Rome: FAO.

Aubinet, M., Grelle, A., Ibrom, A., Rannik, U., Moncrieff, J. Foken, T., … Vesala, T. (2000). Estimates of the Annual Net Carbon and Water Exchange of Forests: The EUROFLUX Methodology. *Advances in Ecological Research*, 30(1), 113-175.

Baird, A., Holden, J. & Chapman, P. (2009). *A Literature Review of Evidence on Emissions of Methane in Peatlands*. Defra, United Kingdom.

Baldocchi, D. (2003). Assessing the eddy covariance technique for evaluating ecosystem carbon dioxide exchange rates of ecosystems: past present and future. *Global Change Biology,* 9, 479-492.

Baldocchi, D., Falge, E., Gu, L., Olson, R., Hollinger, D., Running, S., … Wofsy, S. (2001). FLUXNET: A New Tool to Study the Temporal and Spatial Variability of Ecosystem-Scale Carbon Dioxide, Water Vapour, and Energy Flux Densities. *Bulletin of the American Meteorological Society,* 82(11), 2415-2434.

Ballhorn, U., Siegert, F., Mason, M., & Limin, S. (2009). Derivation of burn scar depths and estimation of carbon emissions with LIDAR in Indonesian peatlands. *Proceedings of the National Academy of Sciences of the United States of America,* 106, pp. 21213–21218.

Battin, T. J., Luyssaert, S., Kaplan, L. A., Richter, A., & Tranvik, L. J. (2009). The boundless carbon cycle. *National Geoscience*, 2, 598–600.

Baum, A., Rixen, T. & Samiaji, J. (2007). Relevance of peat draining rivers in central Sumatra for the riverine input of dissolved organic carbon into the ocean. *Estuarine Coastal and Shelf Science*, 73, 563-570.

Becker, T., Kutzbach, L., Forbrich, I., Schneider, J., Jager, D., Thees, B., & Wilmking, M. (2008). Do we miss the hotspots? The use of very high resolution aerial photographs to quantify carbon fluxes in peatlands. *Biogeosciences,* 5, 1387-1393.

Brady, M A. (1997). *Organic matter dynamics of coastal peat deposits in Sumatra, Indonesia*. Doctoral dissertation. Retrieved from the University of British Columbia Library.

Brown, S., Iverson, L. R., Prasad, A. & Liu, D. (1993). Geographical distributions of carbon in biomass and soils of tropical Asian forests. *Geocarto Int*., 4, 45–59.

Butler, R. A., Koh., L. P. & Ghazoul, G. (2009). REDD in the red: palm oil could undermine carbon payment schemes. *Conservation Letters*, 2, 67-73.

Butler, R. A. & Laurance, W. F. (2009). Is oil palm the next emerging threat to the Amazon? *Tropical Conservation Science*, 2, 1-10.

Carter, C., Finley, W., Fry, J., Jackson, D., & Willis, L. (2007). Palm oil markets and future supply. *European Journal of Lipid Science and Technology*, 109, 307-314.

Center for International Forestry Research (2010). *The local social and environmental impacts of biofuel feedstock expansion: A synthesis of case studies from Asia, Africa and Latin America*. Bogor, Indonesia: German, L., Schnoeveld, G., Skutsch, M., Andriani, R., Obidzinski, K., Pacheco, P., Komarundin, H., Andrianto, A., Lima, M., & Dayang Norawana, A.A.B.

Chimner, R. A. (2004). Soil respiration rates in tropical peatlands of Micronesia and Hawaii. *Wetlands*, 24(1), 51–56.

Chimner, R. A. & Ewel, K. C. (2004). Differences in carbon fluxes between forested and cultivated Micronesian tropical peatlands. *Wetland Ecology and Management*, 12, 419–427.

Corley, R. H. V. & Tinker, P. B. (2003). *The Oil Palm* (4^{th} ed.) Oxford, United Kingdom: Blackwell Science Ltd.

Couwenberg, J. (2009). *Emission factors for managed peat soils (organic soils, histosols): An analysis of IPCC default values*. Ede, the Netherlands: Wetlands International.

Couwenberg, J. (2011). Greenhouse gas emissions from managed peat soils: is the IPCC reporting guidance realistic? *Mires and Peat*, 8, 1-10.

Couwenberg, J., Dommain, R. & Joosten, H. (2010). Greenhouse gas fluxes from tropical peatlands in south-east Asia. *Global Change Biology,* 16, 1715-1732.

Danielsen, F., Beukema, H., Burgess, N. D., Parish, F., Bruhl, C. A., Donald, P. F., … Fitzherbert, E.B. (2009). Biofuel Plantations on Forested Lands: Double Jeopardy for Biodiversity and Climate. *Conservation Biology,* 23(2), 348-358.

Darung, U. (2004, December). The effect of forest fire and agriculture on CO2 emission from tropical peatlands, Central Kalimantan, Indonesia. In A. Iswandi, H.C. Wijaya, S. Guhardja, H. Segah, T. Iwakuma, & M. Osaki (Ed.) *The International Workshop on Human Dimension of Tropical Peatland under Global Environmental Changes*. Bogor, Indonesia.

Delft Hydraulics. (2006). *PEAT-CO2, Assessment of CO2 emissions from drained peatlands in SE Asia (Report Q3943).* Delft, The Netherlands: *Hooijer*, A., Silvius, M., Wösten, H. & Page, S.

Delft Hydraulics. (2009). *Kampar Peninsula Science Based Management Support Project: Summary Interim Report, 2007-2008: First tentative findings on hydrology, water management, carbon emissions and landscape ecology*. Delft, The Netherlands: Hooijer. A., Page, S. & Jauhiainen, J.

Denmead, O. T. (2008). Approaches to measuring fluxes of methane and nitrous oxide between landscapes and the atmosphere. *Plant and Soil,* 309, 5-24.

Desai, A. R., Richardson, A. D., Moffat, A. M., Kattge, J., Hollinger, D., Barr, A., … Stauch, V. J. (2008). Cross-site evaluation of eddy covariance GPP and RE decomposition techniques. *Agricultural and Forest Meteorology*, 148, 821-838.

Di Marco, C., Skiba, U., Weston, K., Hargreaves, K., & Fowler, D. (2004). Field scale N_2O flux measurements from grassland using eddy covariance. *Water, Air & Soil Pollution: Focus,* 4(6), 1567-7230.

Dehue, B., van de Staaij, J., & Chalmers, J. (2009). *Mitigating indirect effects of biofuel production: Case studies and methodology*. Utrecht, The Netherlands: Ecofys.

European Commission Joint Research Center (2010). *Indirect Land Use Change from increased biofuels demand* (EUR 24485 EN). Italy: Publications Office of the European Union.

Fargione, J., Hill, J., Tilman D., Polasky, S., & Hawthorne, P. (2008). Land Clearing and the Biofuel Carbon Debt. *Science,* 319, 1235-1238.

Fitzherbert, E. B., Struebig, M., Morel, A., Danielsen, F., Brühl, C. A., Donald, P. F., & Phalan, B. (2008). How will oil palm expansion affect biodiversity? *Trends in Ecology and Evolution*, 10, 538-545.

Foreign Agricultural Service, United States Department of Agriculture. (July, 2011). Oilseeds: World Markets and Monthly Trade Circular. Retrieved from: http://www.fas.usda.gov/oilseeds/Current/.

Kepong, Selangor: Melling, L., Uyo, L. J., Goh, R., Hatano, R., & Osaki, M (2008). Greenhouse gas fluxes of Loagan Bunut National Park, Sarawak, Malaysia (UNDP/GEF MAL/99/31). Kepong: Forest Research Institute Malaysia.

FLUXNET (2011). *FLUXNET Project*. Retrieved from http://daac.ornl.gov/FLUXNET.

Furukawa, Y., Inubushi, K., Ali, M., Itang, A. M., & Tsuruta, H. (2005). Effect of changing groundwater levels caused by land-use changes on greenhouse gas fluxes from tropical peat lands. *Nutrient Cycling in Agroecosystems*, 71, 81-91.

Germer, J. & Sauerborn, J., (2008). Estimation of the impact of oil palm plantation establishment on greenhouse gas balance. *Environment, Development and Sustainability,* 10, 697-716.

Gibbs, H. K., Johnston, M., Foley, J. A., Holloway, T., Monfreda, C., Ramankutty, N., & Zaks, D. (2008). Carbon payback times for crop-based biofuel expansion in the tropics: the effects of changing yield and technology. *Environment Research Letters,* 3, 1-10.

Gorham, E., (1991). Northern peatlands: Role in the carbon cycle and probably responses to climatic warming. *Ecological Applications*, 1: 182-195.

Hadi, A., Inubushi, K., Furukawa, Y., Purnomo, E., Rasmade, M., & Tsuruta, H. (2005). Greenhouse gas emissions from tropical peatlands of Kalimantan, Indonesia. *Nutrient Cycling in Agroecosystems*, 71, 73-80.

Hadi, A., Inubushi, K., Purnomo, E., Furukawa, Y., & Tsuruta, H. (2002, August). Emission of CH_4 and CO_2 from tropical peatland and factors affecting them. In *The 17th World Congress on Soil Sciences 2002 (WCSS)*. Bangkok, Thailand.

Hadi, A., Inubushi, K., Purnomo, E., Razle, F., Yamakawa, K., & Tsuruta, H. (2000). Effect of land-use change on nitrous oxide (N_2O) emission from tropical peatlands. *Chemosphere - Global Change Science,* 2, 347-358.

Harrison, M. E., Page, S. E., & Limin, S. H. (2009). The global impact of Indonesian forest fires. *Biologist*, 56(3), 156.

Hendricks, D. M. D., van Huissteden, J., Dolman, A. J. and van der Molen, M. K., (2007). The full greenhouse gas balance of an abandoned peat meadow. *Biogeosciences,* 4, 411-424.

Henson., I. E. & Dolmat, M. T., (2003). Physiological analysis of an oil palm density trail on a peat soil. *Journal of Oil Palm Research*, 15, 1-27.

Hergoualc'h, K. & Verchot, V., (2011). Stocks and fluxes of carbon associated with land use change in Southeast Asian tropical peatlands: A review. *Global Biogeochemical Cycles*, 25. Doi:10.1029/2009GB003718.

Hirano, T., Jauhiainen, J., Inoue, T. & Takahashi, H. (2009). Controls on the Carbon Balance of Tropical Peatlands. *Ecosystems,* 12, 873-887.

Hirano, T., Segah, H., Harada, T., Limin, S., June, T., Hirata, R., & Osaki, M. (2007). Carbon dioxide balance of a tropical peat swamp forest in Kalimantan, Indonesia. *Global Change Biology*, 13, 412-425.

Holden, J. (2005). Peatland hydrology and carbon cycling: Why small-scale process matters. *Philosophical Transactions of the Royal Society A,* 263, 2891-2913.

Hooijer, A., Haasnoot, M., van der Vat, M. & Vernimmen, R. (2008). *Master plan for the conservation and development of the ex-mega rice project area in Central Kalimantan, cluster 3: hydrology and peatland water management. Technical report: Peatland subsidence scenarios for the EMRP area.* Deltares, Jakarta: Euroconsult Mott MacDonald.

Hooijer, A., Miettinen, J., Tollenaar, D. and Page, S. E. (in press). *Historical analysis and projection of oil palm plantation extent on peatland in SE Asia; 2010 situation.* Washington, D.C.: International Council on Clean Transportation.

Hooijer, A., Page, S., Canadell, J. G., Silvius, M., Kwadijk, J., Wösten, H., & Jauhiainen, J. (2010). Current and future CO_2 emissions from drained peatlands in Southeast Asia. *Biogeosciences,* 7, 1505-1514.

Hooijer, A., Page, S. E., Jauhiainen, J., Lee, W. A., Idris, A., & Anshari, G. (2011). Land subsidence and carbon loss in plantations on tropical peatland: reducing uncertainty and implications for emission reduction options. *Biogeosciences Discussions* (in press).

Inubushi, K., Furakawa, Y., Hadi, A., Purnomo, E. & Tsuruta, H. (2003). Seasonal changes of CO_2 CH_4 and N_2O fluxes in relation to land-use change in tropical peatlands located in coastal area of South Kalimantan. *Chemosphere*, 52, 603-608.

Inubushi, K. & Hadi, A. (2007). Effect of land-use management on greenhouse gas emissions from tropical peatlands. In Rieley, J. O., Banks, C. J., & Radjagukguk, B. (Ed.) *Proceedings of the International Symposium and Workshop on Tropical Peatland*. Yogyakarta, Indonesia.

Inubushi, K., Otake, S., Furukawa, N., Shibasaki, N., Ali, M., Itang, M., & Tsuruta, H. (2005). Factors influencing methane emissions from peat soils: comparison of tropical and temperate wetlands. *Nutrient Cycling in Agroecosystems*, 71, 93–99.

International Panel on Climate Change. (1997). *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories.* Geneva, Switzerland.

International Panel on Climate Change. (2006). *IPCC guidelines for national greenhouse gas inventories.* Geneva, Switzerland.

Ismail, A. B., Zulkefli, M., Salma, I., Jamaludin, J., & Mohamad Hanif, M. J. (2008). Selection of land clearing technique and crop type as preliminary steps in restoring carbon reserve in tropical peatland under agriculture. In Feehan, J., Farrell, C. A. (Ed.) *Proceedings 13th International Peat Congress*. Tullamore, Ireland.

Jaenicke, J., Rieley, J. O., Mott, C., Kimman, P., & Siegert, F. (2008). Determination of the amount of carbon stores in Indonesian peatlands. *Geoderma*, 147, 151-158.

Jauhiainen, J., Heikkinen, J., Martikainen, P. J. & Vasander, H. (2001). CO_2 and CH_4 fluxes in pristine and peatland converted to agriculture in Central Kalimantan, Indonesia. *International Peat Journal*, 11, 43-49.

Jauhiainen, J., Hooijer, A., & Page, S. E. (2011). Carbon Dioxide Fluxes in an Acacia Plantation on Tropical Peatland. *Biogeosciences Discussions*, 8, 8269-8302.

Jauhiainen, J., Limin, S., Silvennoinen, H. & Vasander, H. (2008). Carbon dioxide and methane fluxes in drained tropical peat before and after hydrological restoration. *Ecology*, 89, 3503–3514.

Jauhiainen, J., Silvennoinen, H., Hämäläinen, R., Kusin, K., Limin, S., Raison, R. J., & Vasander, H. (2011). Nitrous oxide fluxes from tropical peat with different disturbance history and management. *Biogeosciences Discussions*, 8, pp.5423–5450.

Jauhiainen, J., Takahashi, H., Juha, Heikkinen, E. P., Martikainen, P. J., & Vasander, H. (2005). Carbon fluxes from a tropical peat swamp forest floor. *Global Change Biology,* 11, 1788-1797.

Jauhiainen, J., Vasander, H., Heikkinen, J. & Martikainen, P.J. (2002). Carbon fluxes in pristine and developed Central Kalimantan peatlands. In Rieley, J.O. & Page, S.E. (Ed.), *Proceedings of the International Symposium on Tropical Peatlands.* Jakarta, Indonesia.

Jauhiainen, J., Vasander, H., Jaya, A., Takashi, I., Heikkinen, J., & Martiinen, P. (2004). Carbon balance in managed tropical peat in Central Kalimantan, Indonesia. In J. Päivänen, (Ed.), *Proceedings of the 12th International Peat Congress.* Tampere, Finland.

Jauhiainen, J., Vasander, H., Rieley, J., & Page, S. E. (2010). *Tropical Peat Carbon Gas Interaction: Technical report 2.* Leicester, United Kingdom: University of Leicester.

Joosten, H. (2009). *The Global Peatland CO_2 Picture – Peatland status and drainage related emissions in all countries of the world.* Ede, the Netherlands: Wetlands International. Produced for the UN-FCCC meetings in Barcelona, November 2009.

Koh, L. P., & Ghazoul, J. (2008). Biofuels, biodiversity, and people: Understanding the conflicts and finding opportunities. *Biological Conservation*, 141, 2450-2460.

Koh, L. P., Miettinen, J., Liew, S. C. & Ghazoul, J. (2011). Remotely sensed evidence of tropical peatland conversion to oil palm. *Proceedings of the National Academy of Scientists of the United States of America,* 108, 5127-5132.

Koh, L. P. & Wilcove, D. S. (2007). Cashing in palm oil for conservation. *Nature*, 448, 993-994.

Koh, L. P. & Wilcove, D. S. (2008). Is oil palm agriculture really destroying tropical biodiversity? *Conservation Letters,* 1, 60-64.

Kuzyakov, Y. (2006). Sources of CO_2 efflux from soil and review of partitioning methods. *Soil Biology and Biochemistry,* 38, 425-448.

Kyuma, K., Kaneko, N., Zahari, A.B. & Ambak, K. (1992). Swamp forest and tropical peat in Johore, Malaysia. In: Aminuddin B.Y., Tan S.L., Aziz B., Samy J., Salmah Z., Siti Petimah H., Choo S.T., (Ed.), Proceedings of the International Symposium on Tropical Peatland. Kuching, Malaysia.

Laine, A., Sottocornola, M., Kiely, G., Byrne, K. A., Wilson, D., & Tuittila, E. S. (2006). Estimating net ecosystem exchange in a patterned ecosystem: Example from blanket bog. *Agricultural and Forest Meteorology,* 138, 231-243.

Lamade, E. & Bouillet, J. (2005). Carbon storage and global change: the role of palm oil. *OCL - Oleagineux, Corps Gras, Lipides*, 12, 154-160.

Langner, A. & Siegert, F. (2009). Spatiotemporal fire occurrence in Borneo over a period of 10 years. *Global Change Biology*, 15, 48-62.

Lapola, D.M., Schaldach, R., Alcamo, J., Bondeau, A., Koch, J, Koelking, C., & Priess, J. A. (2010). Indirect land-use changes can overcome carbon savings from biofuels in Brazil. *Proceedings of the National Academy of Sciences of the United States of America*, 107, 3388-3393.

Lasslop, G., Reichstein, M., Papale, D., Richardson, A. D., Arneth, A., Barr, A., … Wohlfahrt, G. (2010). Separation of net ecosystem exchange into assimilation and respiration using a light response curve approach: critical issues and global evaluation. *Global Change Biology*, 16, 187-208.

Law, B. & Verma, S. (2004). Introduction. In X. Lee, W. Massman and B. Law (Ed.), *Handbook of Micrometeorology: A guide for surface flux measurement and analysis*. (pp. 1-6). Dordrecht: Kluwer Academic Press.

Lloyd, C.R. (2006). Annual carbon balance of a managed wetland meadow in the Somerset Levels, UK. *Agricultural and Forest Meteorology,* 138, 168-179.

Melling, L., Goh, K.J., Beauvais, C. & Hatano, R. (2007). Carbon flow and budget in a young mature oil palm agroecosystem on deep tropical peat. In: Rieley J.O., Banks C.J., & Radjagukguk B., (Ed.) *Proceedings of the International Symposium and Workshop on Tropical Peatland*, Yogyakarta, Indonesia.

Melling, L., Goh, K.J. & Hatano, R. (2006). Short-term effect of urea on CH_4 flux under the oil palm (*Elaeis guineensis*) on tropical peatland in Sarawak, Malaysia. *Soil Science and Plant Nutrition*, 52, 788–792.

Melling, L., Hatano, R. & Goh, K.J. (2005). Methane fluxes from three ecosystems in tropical peatland of Sarawak, Malaysia. *Soil Biology and Biogeochemistry,* 37, 1445-1453.

Melling, L., Hatano, R. & Goh, K.J. (2005). Soil CO2 flux from three ecosystems in tropical peatland of Sarawak, Malaysia. *Tellus,* 57B, 1-11.

Melling, L., Hatano, R. & Goh, K.J. (2007). Nitrous oxide emissions from three ecosystems in tropical peatland of Sarawak, Malaysia. *Soil Science and Plant Nutrition,* 53, 792-805.

Miettinen, J. & Liew, S.C. (2010). Status of peatland degradation and development in Sumatra and Kalimantan, *AMBIO*, 39, 394-401.

Ministry of Forestry, Government of Indonesia & European Commission, European Union (2001). *Oil palm agriculture in the wetlands of Sumatra: destruction or development?* Italy: Publications Office of the European Union.

Moffat, A.M., Papale, D., Reichstein, M., Hollinger, D. Y., Richardson, A. D., Barr, A. G., … Stauch, V. J. (2007). Comprehensive comparison of gap filling techniques for eddy covariance net carbon fluxes. *Agricultural and Forest Meteorology,* 147, 209-232.

Moore, S., Gauci, V., Evans, C.D. & Page, S.E. (2011). Fluvial organic carbon losses from a Bornean blackwater river, *Biogeosciences*, 8, 901-909.

Murayama, S. & Bakar, Z.A. (1996). Decomposition of Tropical Peat Soils. *Japan Agricultural Research Quarterly,* 30, 153-158.

Murdiyarso, D., Hergoualc'h, K. & Verchot, L.V. (2010). Opportunities for reducing greenhouse gas emissions in tropical peatlands, *PNAS,* 107, 19655-19660.

Neuzil, S.G. (1997). Onset and rate of peat and carbon accumulation in four domed ombrogenous peat deposits, Indonesia. In: J. O. Rieley and S. E. Page (Ed.), *Tropical peatlands* (pp. 55-72). Hereford, United Kingdom: Samara Publishing Limited.

Page, S. E., Hooijer, A., Rieley, J. O., Banks, C. J., & Hoscilo, A. (in press) The Tropical Peat Swamps of Southeast Asia: Human Impacts on Biodiversity, Hydrology and Carbon Dynamics. In: D.J. Gower, J.E. Richardson, B.R. Rosen, L. Rüber and S.T. Williams, eds. *Biotic Evolution and Environmental Change in Southeast Asia*. Cambridge: Cambridge University Press.

Page, S., Hoscito, A., Wösten, H., Jauhiainen, J., Silvius, M., Rieley, J., … Vasander, H. (2009). Ecological Restoration of Tropical Peatlands – Current Knowledge and Future Research Directions. *Ecosystems*, 12, 888-905.

Page, S. E., Rieley, J. O. & Banks, C. J. (2011). Global and regional importance of the tropical peatlands carbon pool. *Global Change Biology,* 17, 798-818.

Page, S., Rieley, J., Shotyk, B. & Weiss, D. (1999). Interdependence of peat and vegetation in a tropical peat swamp forest. *Philosophical Transactions of the Royal Society of London B*, 354, 1885-1897.

Page, S. E., Siegert, F., Rieley, J. O., Boehm, H. D. V., Jaya, A., & Limin, S. (2002). The amount of carbon released from peat and forest fires in Indonesia during 1997. *Nature*, 420, 61-65.

Page, S. E.**,** Wüst, R. & Banks, C. (2010). Past and present carbon accumulation and loss in Southeast Asian peatlands. *PAGES News*, 18(1), 25-26.

Page, S. E., Wüst, R. A. J., Weiss, D., Rieley, J. O., Shotyk, W., & Limin, S.H. (2004). A record of Late Pleistocene and Holocene carbon accumulation and climate change from an equatorial peat bog (Kalimantan, Indonesia): implications for past, present and future carbon dynamics. *Journal of Quaternary Science*, 19, 625-635.

Paoli, G. D., Carlson, K. M., Hooijer, A., Page, S. E., Curran, L. M., Wells, P. L., … Lawrence, D. (2011). Policy Perils of Ignoring Uncertainty in Oil Palm Research: Critique of Koh *et al*. (2011). *Proceedings of the National Academy of Sciences of the United States of America,* 108, E218

Phalan, B. (2009). The social and environmental impacts of biofuels in Asia: An overview. *Applied Energy*, 86, S21-S29.

Ravindranath, N. H., Manuvie, R., Fargione, J., Canadell, J. G., Berndes, G., Woods, J., … Sathaye, J. (2009). Greenhouse Gas Implications of Land Use and Land Conversion to Biofuel Crops. In: R. W. Howarth & Bringezu, S. (Ed.), *Biofuels Environmental Consequences and Interactions with Changing Land Use* (pp. 111-125). New York, NY: Island Press.

Raupach, M.R. & Canadell, J.G. (2010). Carbon and the Anthropocene. *Current Opinion in Environmental Sustainability*, 2, 2010-218.

Reichstein, M. Falge, E., Baldocchi, D., Papale, D., Aubinet, M., Berbigier, P., … Valentini, R. (2005). On the separation of net ecosystem exchange into assimilation and ecosystem respiration: review and improved algorithm. *Global Change Biology*, 11, 1-16.

Reijnders, L. & Huijbregts, M. A. J. (2008). Palm oil and the emission of carbon-based greenhouse gases. *Journal of Cleaner Production,* 16, 477-482.

Renewable Fuels Agency (2010). *Palm Oil Cultivation in Malaysia*. London, United Kingdom: the Stationary Office.

Rieley, J.O. & Page, S.E. (2008, June). Carbon budgets under different land uses on tropical peatland. In Feehan, J., Farrell, C.A. (Ed.), *Proceedings 13th International Peat Congress*. Tullamore, Ireland.

Rinne, J., Riutta, T., Pihlatie, M., Aurela, M., Haapanala, S., Tuovinen, J. P., … Vesala, T. (2007). Annual cycle of methane emission from a boreal fen measured by the eddy covariance technique. *Tellus Series B-Chemical and Physical Meteorology,* 59, 449-457.

Roulet, N. T., Lafleur, P. M., Richard, P. J., Moore, T. R., Humphreys, E. R., & Bubier, J. (2007). Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland. *Global Change Biology,* 13, 397-411.

Rumbang, N., Radjagukguk, B. & Projitno, D. (2008, June). Emission of CO_2 from tropical peat soil under different land use types. In: Feehan, J., Farrell, C.A. (Ed.), *Proceedings 13th International Peat Congress*. Tullamore, Ireland.

Searchinger, T., Heimlich, R., Houghton, R. A., Dong, F. X., Elobeid, A., Fabiosa, … Yu, T. H. (2008). Use of US croplands for biofuels increases greenhouse gases through emissions from landuse change. *Science*, 319, 1238-1240.

Shirakawa, Y., Yokoyama, S. & Imou, K. (2010). Impact of direct land use change process on lifecycle greenhouse gas emissions of palm oil derived biodiesel. *Journal of the Japanese Institute of Energy*, 89, 649-657.

Singh, G. (2008). *Sustainable Development of Peatland for Oil Palm - United Plantations Berhad's experience*. In: Aminuddin B.Y., Tan S.L., Aziz B., Samy J., Salmah Z., Siti Petimah H., Choo S.T., (Ed.), Proceedings of the International Symposium on Tropical Peatland. Kuching, Malaysia.

Stephens, J. C., Allen, L. H. & Chen, E. (1984). Organic soil subsidence. In T. L. Holzer (Ed.), *Man-induced land subsidence: Geological Society of America Reviews in Engineering Geology, v. 6* (pp. 107-122). Boulder, CO: Geological Society of America.

Susila, W. R. (2004). Impacts of CPO Export Tax on Several Aspects of Indonesian CPO Industry. *Oil Palm Industry Economic Journal*, 4, 1-13.

Suyanto, S., Applegate, G., Permana, R., Khususiyah, N., Kurniawan, I. (2004). The role of fire in changing land use and livelihoods in Riau-Sumatra. *Ecology and Society*, 9, 15.

Suzuki, S., Nagano, T. & Waijaroen, S. (1999). Influences of Deforestation on Carbon Balance in a Natural Tropical Peat Swamp Forest in Thailand. *Environmental Control in Biology,* 37, 115-128.

Takahashi, H., Simada, S., Ibie, B., Usup, A., Yudha, & Limin, S. H. (2002, August). Annual changes of water balance and a drought index in a tropical peat swamp forest of Central Kalimantan, Indonesia. In Rieley, J. O. & Page S. E. (Ed.), *Proceedings of the International Symposium on Tropical Peatlands*. Jakarta, Indonesia.

Takakai, F., Morishita, T., Kuramochi, K., Darung, U., Dohong, S., Limin, S. H., & Hatano, R., (2005, December). The effect of forest fire and agriculture on CH_4 and N_2O emission from tropical peatlands, Central Kalimantan, Indonesia. In: Iswandi, A., Wijaya, H. C., Guhardja, S., Segah, H., Iwakuma, T. & Osaki, M. (Ed.), *Proceedings of the International Workshop on Human Dimension of Tropical Peatland under Global Environmental Changes*. Bogor, Indonesia.

Takakai, F., Morishita, T., Hashidoko, Y., Darung, U., Kuramochi, K., Dohong, S., ... Hatano, R. (2006). Effects of agricultural land-use change and forest fire in N_2O emission from tropical peatlands, Central Kalimantan, Indonesia. *Soil Science and Plant Nutrition*, 52, 662-674.

Taylor, D. & Ali, M. (2001). *Biogeochemical responses to land cover changes in coastal peatland catchments: spatial and temporal fluxes in greenhouse gas emissions and peat subsidence, Jambi Province, Sumatra*. Dublin, Ireland: Southeast Asia Regional Committee for START and United Nations Office for Project Services

Tan, K.T., Lee, K.T., Mohamed, A.R. & Bhatia, S. (2009). Palm oil. Addressing isues and twoards sustainable development. *Renewable and Sustainable Energy Reviews*, 13(2), 420-427.

Teh, Y. A., Silver, W. L., Sonnentag, O., Detto, M., Kelly, M., & Baldocchi, D. D. (2011). Large greenhouse gas emissions from a temperate peatland pasture. *Ecosystems*, 14: 311-325.

Terry, R. E., Tate, R. L., & Duxbury, J. M. (1981). The effect of flooding on nitrous oxide emissions from an organic soil. *Soil Science*, 132, 228–232.

Ueda, S., Go, C. S. U., Yoshioka, T., Yoshida, N., Wada, E., Miyajima, T., … Boonprakum, S. (2000). Dynamics of dissolved O2, CO2, CH4, and N2O in a tropical coastal swamp in southern Thailand. *Biogeochemistry*, 49, 191–215.

van der Werf, G. R., Dempewolf, J., Trigg, S. N., Randerson, J. T., Kasibhatia, P. S., Giglio, L., Murdiyarso, D., Peters, W., Morton, D. C., Collatz, G. J., Dolman, A. J., & DeFries, R. S. (2008). Climate regulations of fire emissions and deforestation in equatorial Asia. *Proceedings of the National Academy of Sciences of the United States of America*, 105, 20350-20355.

Van Kraalingen, D. W. G., Breure, C. J. & Spitters, C. J. T. (1998). Simulation of oil palm growth and yield. *Agricultural and Forest Meteorology*, 46, 227-244.

Venter, O., Meijaard, E., Possinghma, H., Dennis, R., Sheil, D., Wich, S., … Wilson, K. (2009). Carbon payments as a safeguard for threatened tropical mammals. *Conservation Letters*, 2, 123-129.

Verwer, C., van der Meer, P. & Nabuurs, G.(2008). *Review of carbon flux estimates and other greenhouse gas emissions from oil palm cultivation on tropical peatlands - Identifying the gaps in knowledge*. Wageningen, the Netherlands: Alterra.

Vien, D. M., Phuong, N. M., Jauhiainen, J. & Guong, V. T. (2007, August). Carbon dioxide emission from peatland in relation to hydrology, peat moisture, humification at the Vodoi National Park, Vietnam. In: Rieley, J.O., Banks, C.J. & Radjagukguk, B. (Ed.), *Proceedings of the International Symposium and Workshop on Tropical Peatland*. Yogyakarta, Indonesia.

Wang, W., Dalal, R. C., Reeves, S. H., Butterbach-Bahl, K., & Kiese, R. (2011). Greenhouse gas fluxes from an Australian subtropical cropland under long-term contrasting management regimes. *Global Change Biology*. doi: 10.1111/j.1365-2486.2011.02458.x

Wicke, B., Dornburg, V., Junginger, M. & Faaij. A., (2008). Different palm oil production systems for energy purposes and their greenhouse gas implications. *Biomass and Bioenergy*, 32, 1322-1337.

Wösten., J. H. M. & Ritzema, H. P., (2001). Land and waer mangement options for pealtland development in Sarawak, Malaysia. *International Peat Journal*, 11, 59-66.

Wösten, J. M. H., Ismail, A. B. & van Wijk, A. L. M., (1997). Peat subsidence and its practical implications: a case study in Malaysia. *Geoderma,* 78, 25-36.

Wösten, J. H. M., Clymans, E., Page, S., Rieley, J., & Limin, S. (2008). Peat-water interrelationships in a tropical peatland ecosystem in Southeast Asia. *Catena,* 73, 212-224.

Wüst, R. A. J. & Bustin, R. M. (2004). Late Pleistocene and Holocene development of the interior peat-accumulating basin of tropical Tasek Bera, Peninsular Malaysia. *Palaeogeography, Palaeoclimatology, Palaeoecology*, 211, 241-270.

Yeh, S., Jordaan, S. M., Brandt, A. R., Turetsky, M. R., Spatari, S., & Keith, D. W. (2010). Land use greenhouse gas emissions from conventional oil production and oil sands. *Environmental Science & Technology*, 44, 8766-8772

Yew. F. K., Sundram, K. & Basiron, Y. (2010). Estimation of GHG emissions from peat used for agriculture with special reference to oil palm. *Journal of Oil Palm and the Environment*, 1, 17-25.

Yule, C. M. & Gomez, L. N., (2009). Leaf litter decomposition in a tropical peat swamp forest in Peninsular Malaysia. *Wetlands Ecology and Management*. 17, 231-241.

Ywih, C, H., Ahmed, O. H., Majid, N. M. A. & Jalloh, M. B. (2010). Effects of Converting Secondary Forest on Peat to Oil Palm Plantation on Carbon Sequestation. *American Journal of Agricultural and Biological Scieince*, 4, 123-130.

Zhou, A. & Thompson, E. (2009). The development of biofuels in Asia. *Applied Energy*, 86, S11-S20.

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