Reports from the Technical Panels of the 2nd Greenhouse Gas Working Group of the Roundtable on Sustainable Palm Oil (RSPO)

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OVERVIEW

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In November 2008, the RSPO Executive Board established a Green House Gas Working Group (GHG WG) that was charged with reviewing relevant information on palm oil production and GHG emissions. The GHG WG was convened to inform discussions within the RSPO as to the sources and dimensions of GHG emissions from the palm oil supply chain and the need to respond to text within the RSPO Principles and Criteria (P&C) that refer to GHG emissions.[1] The GHG WG was charged to review the current criteria of the P&C and make recommendations to the Executive Board regarding options for reducing GHG emissions from the palm oils supply chain. Special emphasis was placed on understanding GHG emissions from the development of new plantations, because these are widely considered to be the greatest source of GHG emissions by the sector due to deforestation and other forms of land use change. Similarly, documenting the dimensions of GHG emissions from plantations on peat soils were highlighted as important, because of recent scientific reports that those emissions were a major source of GHG emissions in both Indonesia and Malaysia.

The participants of the GHG WG agreed on many issues, specifically the potential to reduce emissions by improving efficiencies in the use of fossil fuels and fertilizers, increasing the use of biomass energy, avoiding biomass burning in the establishment of new plantations, and capturing methane emissions from palm oil mill effluent treatment ponds (POME). The GHG-WG was not able to reach consensus, however, on the dimensions of net GHG emissions from land-use change or from operations of plantations on peat soils. Subsequently, the Executive Board of the RSPO determined that it was necessary to convene a second working group to complete the process. Referred to as the GHG-WG2, this commissions organized its activities into six work streams, each of which focused on specific tasks, an approach that separated the technical issues from policy and market discussions (Box 1). This effort succeeded in reaching the much sought after consensus and the GHG WG2 made a series of recommendation to the RSPO Executive Board. These recommendations were forwarded to a P&C Task Force that was conducting the first technical review of the certification standard, which was originally approved in 2005. The P&C Task Force considered the recommendations, which called for producers to begin monitoring and reporting their emissions, as well as to reduce emissions from existing operations and future expansion of new plantations (Table 1).

The ability to monitor emissions, as well as taking actions to reduce them, is very much dependent on having access to objective information that is both accurate and precise. The first three work streams of the GHG WG2 were convened specifically to address the lack of objective information; as such, they constitute technical panels that provide information to the RSPO community. The composition of the panels was carefully considered and arrived at by consensus to ensure their technical and scientific competence, as well as a balanced perspective regarding the issues being evaluated. For example, the LCA Panel included experts from both the downstream and upstream sectors of the palm oil supply chain, including representatives from corporations, academics and independent consultants, while the Peat Land Working Group included both ecologists and plantation managers. In the case of the Science Panel, all the participants had advanced degrees in ecology, forestry, soil science or geography, and included scientists from Southeast Asia, Europe and North America.

[1] Principle 5: Environmental responsibility and conservation of natural resources and biodiversity Criterion 5.6 Plans to reduce pollution and emissions, including greenhouse gases, are developed, implemented and monitored.

Box. 1.

The six work streams of the RSPO GHG WG 2.

Work stream #1: A group of technicians skilled in Life Cycle Analysis (LCA) developed a framework and informatics tool to facilitate RSPO producer members seeking to measure, monitor and report GHG emissions within their operations, including palm oil mills, corporate estates and smallholder suppliers.

Work stream #2: A multi-stakeholder commission composed of ecologists and plantation managers participating as the Peat Land Working Group reviewed current management practices to develop recommendations for better management practices for existing oil palm plantations operating on peat and evaluated options for the rehabilitation of degraded peat lands, as well as review the scientific and technical literature to document the environmental and social impacts from oil palm plantations operating on peat soils.

Work stream #3: External scientists were commissioned to review the scientific literature on carbon stocks and emissions factors for different types of greenhouse gases and to document land use change in the major palm oil producing regions of Southeast Asia. This information was used to estimate historical CO_2 emissions linked to the palm oil sector and to model the future emissions based on different regulatory and market scenarios.

Work stream #4: The full membership of the GHG WG2 met periodically to evaluate how market innovation or regulatory reform can motivate RSPO members to reduce their GHG emissions, including carbon markets and other forms of climate finance that might promote land swaps, improve smallholder productivity and integrate HCV set asides into GHG accounting protocols.

Work stream #5: The full membership of the GHG WG2 focused its attention on the downstream sector of the palm oil supply chain in response to concerns that too-much emphasis was being placed on the producer members of the RSPO and that retailers, consumer goods manufacturer, banks and NGOs also had a responsibility to lower their GHG emissions profiles.

Work stream #6: A group of RSPO producer members highlighted the voluntary actions they were taking to reduce GHG emissions. This included investing in methane capture technology to reduce emissions from palm oil mill effluent (POME), expand extension services for smallholders, and using low carbon criteria for selecting landscapes for future expansion of oil palm estates.

Table 1. RSPO Principles & Criteria that specifically refer to greenhouse gases

Principle 5: Environmental responsibility and conservation of natural resources and biodiversity	
Criteria 5.6 Plans to reduce pollution and emissions, including greenhouse gases, are developed, implemented and monitored	Indicators: 5.6.1 An assessment of all polluting activities shall be conducted, including gaseous emissions, particulate/soot emissions and effluent. 5.6.2 Significant pollutants and greenhouse gas (GHG) emissions shall be identified, and plans to reduce or minimize them implemented. 5.6.3 A monitoring system shall be in place, with regular reporting on progress for these significant pollutants and emissions from estate and mill operations, using appropriate tools.
Principle 7: Responsible development of new plantings	
Criteria 7.8 New plantation developments are designed to minimize net greenhouse gas emissions.	Indicators: 7.8.1: The carbon stock of the proposed development area and major potential sources of emissions that may result directly from the development shall be identified and estimated. 7.8.2: There shall be a plan to minimize net GHG emissions which takes into account avoidance of land areas with high carbon stocks and/or sequestration options.

The results of all three technical panels were concluded prior to the final meeting of the GHG WG2 in November of 2011, but the publication of the final reports required several more months of analysis and writing, which was followed by a peer review that led to additional modifications and, finally, an editorial revision to ensure consistency in the use of terms, acronyms and style. The RSPO Secretariat has published two manuals on best practices for the operation of oil palm plantations on peat (Lim *et al.,* 2012) and the restoration of peat swamps (Faisal *et al.,* 2012), both of which are freely available as e-books on the RSPO web page. Two more papers from the Peat Land Working Group are included in this publication, both of which are reviews of the scientific literature (see below)

The LCA Panel has organized their findings into three publications, all of which examine or support the PalmGHG calculator, an informatics tool they developed to assist producers monitor the GHG emissions from the operations of palm oil mills, as well as the establishment and management of oil palm plantations. The tool was designed to document the GHG emissions in the supply chain, in order to identify points where emissions are large (so-called GHG hotspots) and predisposed to interventions that can reduce those emissions. As such, it allows the producer to monitor emissions for all three major GHG (CO_2, CH_4 and N_2O) from all major sources, which includes: land clearing for new plantations, operations on peat soils, fertilizer use, management of palm oil mill effluent (POME), and the consumption of fossil fuels for transport and industrial operations. Similarly, the tool incorporates modules to document biomass sequestration from the plantation and conservation set asides, as well as to quantify avoided emissions from the sale of biomass energy that displaces fossil fuel use elsewhere. Since the tool is customized for the industry, it summarizes the information as tones of CO_2 equivalents (CO_2e) per hectare of plantation and per ton of product, which includes both crude palm oil (CPO) and crude palm kernel oil (CPKO).

Two of the publications written by the LCA Panel have been submitted to peer-reviewed journals: One focuses on the mathematical models and biophysical assumptions that underpin the PalmGHG calculator (Bessou *et al.,* 2013 - in review), while the second paper evaluates the results from several contrasting case studies that demonstrate how the tool can be used to stratify net GHG emissions by source and type (Bessou *et al.,* 2013 - in preparation). The third publication is a manual written for individuals who wish to use the PalmGHG calculator to monitor the GHG emissions of an operating palm oil mill and the associated oil palm plantations (Chase *et al.,* 2012).

In this publication, we present seven papers that address emissions from land use change due to the expansion of the palm oil sector and from the cultivation of oil palm plantations on peat soils in Southeast Asia. The first five papers are the final reports produced by the Science Panel and have been organized to reflect the methodological approach used to estimate the historical emissions of CO_2, the most significant GHG linked to land use change, and generate the information resources required to project potential emissions into the future.

The first paper provides a review of the scientific literature covering estimates of above and below ground carbon stocks for a broad range of land use types in Southeast Asia (Agus *et al.,* 2013a – this publication). Net emission factors are calculated by subtracting the time-averaged carbon stock before and after land use change, which allows the authors to generate emission factors for all potential types of land use change involving any one of the 22 different land cover classes included in the study. This methodology automatically incorporates the carbon sequestered by oil palm plantations into emission factor calculations, which can lead to negative emission values (e.g., sequestration) when plantations are established on shrub or grassland landscapes. Mean carbon stock values derived from the values included within this review were used to provide an objective estimate for each emission factor, including those specifically linked to the palm oil sector, but other forms of land use change linked to intensive agriculture or when forest is degraded due to logging and wildfire were also estimated using the same methodology. Finally, this paper provides a review and discussion of the uncertainties linked to estimating emission from oil palm plantations on peat, including those from peat fires and from the degradation or oxidation of peat soils subject to partial drainage.

In the second paper, Gunarso *et al*. (2013 – this publication) report on a comprehensive and original study that used satellite images to document land use change over the last two decades for the major palm oil producing regions of Southeast Asia. These include Peninsular Malaysia, Sabah and Sarawak in the Malaysia Federation, Sumatra and Kalimantan and Papua in the Republic of Indonesia, and Papua New Guinea. Like the first chapter, this paper is organized around 22 land cover classes and seeks to document how land use has shifted among these categories across three temporal epochs: 1990 to 2000, 2001 to 2005 and 2006 to 2010. This decision to focus on 22 distinct categories presented a challenge to the technicians who conducted the work, but it allowed the authors to quantify the different types of land cover that were converted to oil palm plantations (Box 2). Moreover, the stratification of land cover and temporal period allowed the authors to track how land use change varied over time and among sub-regions for the palm oil sector, as well as for other productive sectors, such as forest exploitation, intensive agriculture, and agroforestry. In addition, the authors document the expansion of oil palm plantations on peat soils using a combination of satellite images and soil maps, which created an essential data source for estimating the total historical CO_2 emission from the palm oil industry.

Box 2.

Major findings on oil palm plantations land use change (Gunarso et al. 2013 – this publication)

- Forest conversion as a source of land for new plantations between 1990 and 2010 was documented at 3.5 Mha or 37% when summed over all regions.
- Less than 4% of oil palm plantations originated from the conversion of undisturbed forest.
- Forest conversion was important in Kalimantan (44%), Papua (61%), Sarawak (48%) and Sabah (62%).
- The conversion of agroforest and rubber plantations was important in Sumatra (59%) and Peninsular Malaysia (44%).
- Deforestation as a source of land for expansion of oil palm varied from 48% between 1990 and 2000 to about 20% between 2001 and 2005 and 36% between 2006 and 2010.

The data sources from the first two papers are integrated in the third paper (Agus *et al.,* 2013b – this publication), which combines the emissions factors calculated by Agus *et al.,* (2013a – this publication) with the land use change data produced by Gunarso *et al.,* (2013 – this publication). Essentially, the land use change matrix was multiplied by a corresponding matrix of emission factors to generate a matrix of net CO_2 emissions (or sequestration). This exercise was conducted separately for each sub-region and each temporal period to generate CO_2 emissions profiles that were aggregated into larger categories to facilitate the communication of results. The presentation of the results focus on tracking the growth of emissions linked to the palm oil sector over time, which highlights the variability of emissions from forest conversion and peat fires, but the consistent increase in emissions from peat oxidation (see Box 3). The decision to include other forms of land use change in the study allowed the authors to compare the emissions from the expansion and operations of oil palm plantations with other forms of land use, most notably the sequential degradation of forest by logging and wildfire (see Box 4).

The results from all three papers set the stage for the fourth paper (Harris et al., 2013 – this publication), which projects both land use and CO_2 emissions between 2010 and 2050 for three contrasting scenarios, all of which are based on the supposition that total production of palm oil will double by 2050.

Box 3.

Major findings on CO2 emissions from oil palm plantations (Agus et al., 2013b – this publication).

- Oil palm plantations on peat represent 18% of the spatial footprint of palm oil, but contribute 64% of total emissions linked to land use by 2010.
- There are two sources of emission from peat: peat fires (16% of total emissions between 2006 and 2010) and the oxidation of peat due to drainage (48% of total emissions between 2006 and 2010)
- Emissions from deforestation and peat fires are one-time events and decisions on where or how new plantations are established will immediately impact GHG emissions.
- Emissions from peat oxidation are recurrent emissions that will occur continuously until the plantation is abandoned and the soils are re-wetted.

Box 4.

Major findings on emissions from palm oil in the context of other land uses (Agus et al., 2013b – this publication).

- GHG emission from land use change and peat oxidation between 2006 and 2010 showed that the palm oil sector contributed 16% of total emissions from land use and land use change in Indonesia and 32% in Malaysia.
- In Indonesia, the largest source of historical CO_2 emissions from land use change was due to forest degradation (40%), either from the transition from undisturbed forest to disturbed forest due to logging or from the degradation of degraded forest to shrub land by wildfire.
- Emissions from peat oxidation from degraded swamp forest impacted by logging and drainage are greater than the emissions from oil palm plantations on peat, representing about 22% of total emissions in Indonesia and 13% in Malaysia when all forms of land and land use change are considered.

One of those scenarios, referred to as Business as Usual, assumes the future will be like the past with increased supply coming from the spatial expansion of the industry. The second scenario assumes that demand for land will be diminished due to productivity improvements and that future expansion will be shifted away from high carbon landscapes and is referred to as

Moratorium on Peat. The third scenario is similar to the second scenario, but incorporates an additional assumption that existing plantations on peat soils will be removed from production over time and those lands will be restored as a quasi-natural peat swamp. The objective of the scenario exercise is neither to predict the future nor recommend one development pathway over another, but to visualize what the impact of different development strategies might have on the carbon footprint of the industry when visualized over several decades.

The fifth paper presents the land use change study for Malaysia prepared by the Malaysian members of the Science Panel (Rashid *et al.,* 2013 – this publication), which was completed prior to the region wide study that likewise includes Malaysia, as well as Indonesia and Papua New Guinea (Gunarso *et al.,* 2013 – this publication). The results of the two studies are broadly similar, especially for the palm oil sector which was the principal focus of both research efforts. The Malaysian study was compiled using existing datasets that recognized only seven land cover categories and two distinct temporal periods, while the Indonesian and Papua New Guinea data sets comprised 22 categories. The different spatial and temporal attributes of the two studies made a region wide comparison impossible. Consequently, the decisions was made by the GHG WG2 to expand the studies undertaken by Gunarso *et al.* (2013 – this publication) in order to create a region wide dataset for the parallel research efforts under way to document historical emissions (Agus *et al.,* 2013b – this publication) and project future emissions (Harris *et al.,* 2013 – this publication).

The sixth and seventh studies papers are literature reviews commissioned by the Peat Land Working Group, which describe the environmental and social impacts of oil palm plantations operating on peat soils (Schrier-Uijl *et al.,* 2013 – this publication) and an overview of the various methodologies available for monitoring GHG emissions from oil palm plantations (Schrier-Uijl & Anshari, 2013 – this publication). The abbreviated discussion on peat provided by Agus *et al.* (2013a – this publication) is complementary to the more comprehensive and detailed review provided by Schrier-Uijl *et al.* (2013 – this publication). The two papers come to different conclusions as to the dimensions of CO_2 emissions from both peat fires and peat oxidation, which reflect differences in methodological approaches and data interpretation that are not uncommon within the scientific community.

These studies were commissioned to inform an ongoing discussion within the RSPO about GHG and palm oil; hopefully, it will also shed light on a larger discussion that includes policy makers, academics and the general public. When these studies were first commission, we (the editors) were surprised that there had been no systematic evaluation of the impact of the palm oil sector on land use change and GHG emissions. As the three literature reviews show, there are numerous research articles and efforts that focus on deforestation, but these studies have not adequately segregated the impact of oil palm from other drivers of land use change. Similarly, there is an abundance of information on the expansion of oil palm plantations, but this information has not adequately identified the many different land cover types that have been, and are being, converted to oil palm plantations. Although it is widely known that intensive logging and wildfire are major sources of forest degradation, few studies had related this phenomenon with the land use trajectory that causes forest to be degraded, sometimes severely so, prior to its conversion to oil palm plantation. Those few studies that did attempt to understand all or many of these issues were focused on a specific landscape or sub-region.

The four integrated studies published here provide the first sector wide evaluation of the CO_2 emissions for the palm oil sector that spans two decades for the major palm oil producing regions in Southeast Asia – where more than 85% of this globally important commodity is produced. Like almost all research endeavours, these studies do not resolve all of the issues related to oil palm and greenhouse gases, but hopefully, they provide a foundation that will support industry's ongoing efforts to reduce the GHG emissions from the palm oil supply chain.

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REVIEW OF EMISSION FACTORS FOR ASSESSMENT OF CO_2 EMISSION FROM LAND USE CHANGE TO OIL PALM IN SOUTHEAST ASIA

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ABSTRACT

This paper reviews published reports of carbon stocks, emission factors and approaches for estimating CO_2 emissions from land use change and peat soils. Above ground carbon stock values were based on studies representative of major land cover types for Indonesia, Malaysia and Papua New Guinea and include undisturbed upland forests, undisturbed swamp forest, disturbed upland forests, disturbed swamp forest, shrub land and swamp shrub land, with average above ground carbon stock values of 189, 162, 104, 84, 30 and 28 Mg C ha^{-1}, respectively. The time-averaged above ground carbon stock for oil palm plantations, rubber plantations, timber plantations, mixed tree crops (agroforest) and agricultural crop land was estimated at 36, 56, 44, 54 and 11 Mg C ha^{-1}, respectively. The emissions factors linked to land use change among these land cover types is the difference in carbon stocks between any two of these values converted to Mg CO_2 ha^{-1}.

Emissions from the oxidation of peat soils can be estimated by measuring the amount of CO_2 released from the soil surface over discrete time periods (closed chambers), or from the net changes of soil carbon measured over one or several time periods (subsidence studies). Emissions factors are expressed in Mg of CO_2 per unit area per unit of time (Mg CO_2 ha^{-1} yr^{-1}) and vary between 20 to 95 Mg CO_2 ha^{-1} yr^{-1} due to natural variability and disturbance, as well as to uncertainties in the methodological protocols used to measure or model emissions. We recommend 43 Mg CO_2 ha^{-1} yr^{-1} as a time-averaged default value for estimating emissions caused by the oxidation of peat for oil palm plantations operating on peat soils that have a mean water table depth of 60 cm.

Emissions from fires that impact peat soils when used to clear vegetation during plantation establishment vary depending on weather conditions, and can range from zero in wet years to up to more than 50 cm deep during extreme drought linked to *El Niño* events. We recommend using an average value of 15 cm depth of burnt peat soils for estimating emissions from plantations established on forest landscapes and 5 cm depth when clearing shrub land. Emissions from peat fires are similar to those from land use change, because both are one-time emissions generated while establishing a new plantation. In contrast, emissions from the oxidation of peat recur annually throughout the life time of a plantation that operates on partially drained peat soils.

Keywords: *land cover, land use change, carbon stock, above ground biomass, emissions factor, soil carbon, peat, peat oxidation, fire*.

INTRODUCTION

The rapid expansion of oil palm over the past two decades has led to the transformation of large areas of forest and plantation landscapes throughout Southeast Asia and is believed to be one of the major sources of greenhouse gas (GHG) emissions linked to land use in the region (Agus *et al.,* 2010, Ekadinata & Dewi, 2011, Wicke *et al.,* 2011). Demand for palm oil continues to grow and the sector continues to invest in expanded production through multiple strategies, including by increasing yield and avoiding waste, but also by expanding the area under cultivation.

The ongoing and future expansion of oil palm plantations may, or may not, result in future emissions of CO_2, the most significant GHG linked to land use, depending on the type of land cover that is converted for new plantations. For example, if expansion occurs on forest landscapes with high above- and below-ground carbon stocks, then net emissions linked to the sector will be proportionally large. In contrast, if the source of land for new plantations has low C stock value, such as shrub land or agroforest, then future expansion could be considered carbon neutral. In some cases, expansion might actually be carbon positive if the initial carbon stock is less than that of oil palm as is the case with grassland and most types of annual crops.

In addition to land cover change, the conversion and drainage of peat soils creates an additional source of CO_2 emissions (Wösten *et al.,* 2008; Hooijer *et al.,* 2010; Page *et al.,* 2011a; Parish *et al.,* 2007). A major component of emissions originating from peat formations is the result of fire used as a management tool when establishing new plantations; however, CO_2 is also released via anaerobic decomposition once the anoxic conditions of the peat soil profile are modified to facilitate the cultivation of oil palm. Peat swamps form when input from photosynthesis is greater than decomposition leading to the accumulation of partially decayed organic matter (e.g., peat); drainage reverses this equilibrium leading to a gradual decline in the amount of peat stored in the soil. Water management is an important factor in determining the level of CO_2 emissions from oil palm plantations operating on peat soils and has direct implications locally in the form of peat subsidence, which increases susceptibility to floods and droughts, and affects the general environment in the form of CO_2 emission and loss of biodiversity. Emissions caused by the oxidation of peat are recurrent and will continue until the plantation is removed from production and re-flooded to create the anoxic conditions that favor peat formation.

The absolute and relative magnitude of CO_2 emissions from land use change and the conversion of peat soils have been subject to much speculation and vigorous debate because of the uncertainty and variability associated with published reports. This paper provides a review of the scientific and technical literature in order to provide representative values for general use and explains the method of emission calculation associated with land use changes.

METHODOLOGY OF EMISSION CALCULATION

Net emission from land use and land use changes can be estimated based on equations provided by IPCC (2006):

$$\Delta C = \Sigma \text{ (Activity data * Emission factor)} \qquad [1]$$

Where ΔC is the change in carbon stock, *Activity data* is the area undergoing a specific type of land use change that emits carbon, and *Emission factor* is the total loss of carbon stock per unit land area during the specific type of land use change. Carbon emissions can be expressed in terms of C loss or can be converted to CO_2 by multiplying with a factor of 44/12 which is the molecular weight of CO_2 per unit atomic weight of C. If the activity data account for all possible land use changes within a classification system, equation [1] can be rewritten as:

$$\Delta C = \sum_{ij} A_{ij} \, [\Delta C_{ij\,LB} + \Delta C_{ij\,DOM} + \Delta C_{ij\,SOIL}] \,/ T_{ij} \qquad [2]$$

Where

ΔC = the change in all carbon pools in a unit of time

A_{ij} = the activity data or area of land use under land cover type *i* that change to type *j* during an observation period

$\Delta C_{ij\,LB}$ = change in carbon stock in the living biomass (above + below ground)

$\Delta C_{ij\,DOM}$ = change in carbon stock in dead organic matter, especially dead vegetation (above + below ground)

$\Delta C_{ij\,SOIL}$ = the change in carbon stock in the soil

T_{ij} = the length of the observation period and time scale of calculation

The living biomass (LB), dead organic matter (DOM) or necromass, and soil organic matter (SOIL) are the main carbon pools. There are more published emission data for living biomass and soil but below ground biomass and necromass are rarely assessed (Hairiah *et al.,* 2001). Secondary forests and newly planted agricultural lands may have high amounts of above ground necromass (Hairiah & Rahayu, 2007), but this decomposes on the ground within a few years resulting in a lower C stock when time-averaged. Due to the few data available necromass is not included in the national or sub-national calculations shown in Agus *et al*. (2013 – this publication).

Carbon in above ground biomass and in necromass together constitutes the total above ground carbon stock. The below ground biomass can be estimated from root/shoot ratios. Default values for the root/shoot ratio of tree biomass are around 1/4. However, the ratio varies depending on species, soil and climatic conditions (Hairiah & Rahayu, 2007).

CARBON STOCK ESTIMATES

There is a wide range of estimates in the literature of carbon stock in plant biomass and we provide a review of those values for the 22 land cover types used in the companion studies (Table 1: see Gunarso *et al.,* 2013 and Agus *et al.,* 2013, this publication). The sources mainly include only the carbon in above ground biomass as there is very little reliable data for below ground biomass and soil organic matter for most land cover types, a data deficiency that is compounded by very high levels of natural variability in both natural and human altered ecosystems. Carbon stock estimates for undisturbed natural vegetation types represent values from habitats assumed to be at equilibrium and, as such, are effectively equivalent to time-averaged values; however, values from disturbed habitat types represent their status at the time of conversion and are not equivalent to a time-averaged value. Values for all human altered categories, such as oil palm, rubber plantations, timber and pulp plantations, agroforest and intensive agricultural are time-averaged values that reflect the life cycle of individual production systems.

Above Ground Biomass

Published reports on forest carbon have evolved over time. Early papers tended to have relatively high estimates of plant biomass carbon stock in undisturbed forest, while more recent ones tend to have much lower estimates as the scientific community has become more interested in the global carbon cycle and the impact of disturbance on ecosystem function. For example, Palm *et al*. (1999) estimated carbon stocks in the plant biomass of primary (undisturbed) forest that ranged from 207 to 405 Mg C ha^{-1}, while secondary (disturbed) forest in Kalimantan stores between 58 to 203 Mg C ha^{-1} (Brearly *et al.*, 2004; Rahayu *et al.*, 2005; Harja *et al.*, 2011). Laumonier *et al.* (2010) working in South Sumatra found above ground forest carbon stocks to be between 135-240 Mg C ha^{-1}, with an average of 183 Mg C ha^{-1}. Most of these estimates were based on the non-destructive measurement of tree girth with reference to a wood density database maintained at the World Agroforestry Centre (ICRAF), resulting in tree biomass and carbon stock estimates based on only one of a few allometric relationships. The estimates of Harja *et al.* (2011) used the allometry of Chave *et al.* (2005) which is more conservative compared to those of Basuki *et al.* (2009), Brown *et al.* (1989) and Ketterings *et al.* (2001).

A recent study derived from the National Forest Inventory of Indonesia, covering more than 2000 forest plots scattered across the country and stratified by ecological zone. has provided a significantly, and surprisingly, lower estimate of average forest C stock, ranging from 93 Mg ha^{-1} for undisturbed forests to 74 Mg ha^{-1} for low density disturbed forests (Ekadinata & Dewi, 2011; Harja *et al.*, 2011). The level of replication for undisturbed forest, however, was lower than that for other types of forest cover, and quality control of forest inventory data, required by allometric equations that depend on wood density, may be insufficient. Consequently, we recommend using mean values from all listed results (Table 1). Estimates for rubber plantations ranged from 25 to 143 Mg C ha^{-1} (Ziegler *et al.*, 2011) with a mean time-averaged estimate of 56 Mg C ha^{-1}. Estimates for timber and pulp plantations (Table 1) are lower due to the shorter life cycle that characterizes that industry, while mixed tree crops or agroforest landscapes are highly heterogeneous, reflecting age of settlements and population density.

For oil palm plantations, the carbon stock data are surprisingly variable considering the oil palm is a tree with relatively simple allometry and is cultivated in uniform stands comprised of equal age cohorts. Differences occur largely due to the assumptions and components included in the modeling or measurement protocol, with only some studies including persistent leaf bases, dead fronds (e.g., necromass), ground cover and roots. On average, necromass on the surface will decompose within 12-18 months (Khalid *et al.*, 2000) and, in some cases, may increase soil carbon stock (Mathews *et al.*, 2010; Haron *et al.*, 1998) and nutrient supply (Chiew & Rahman, 2002; Salétes *et al.*, 2004). If data are provided for necromass, however, the decomposition rate should be taken into account when calculating time-averaged necromass stock; otherwise, the accounting for the decomposing necromass will result in double accounting in a carbon stock assessment. Estimates of time-averaged above ground carbon stock for oil palm range from 23 to 60 Mg C ha^{-1}. We recommend using the mean value of 36 Mg C ha^{-1} (Table 1).

Table 1. Above ground carbon stocks (AGC) of different land use classes. Estimates for undisturbed natural vegetation types represent values from habitats assumed to be at equilibrium, while values from disturbed habitat types represent their status at the time of conversion. Values for all human altered categories, such as oil palm, rubber plantations, timber and pulp plantations, agroforest and intensive agricultural are time-averaged values that reflect the life cycle of individual production systems. Unless otherwise stated, data are for above ground biomass only and were obtained in Indonesia.

Land use type and description [(1)]	AGC (Mg ha^{-1})	Reference; remarks
UNDISTURBED UPLAND FOREST Natural forest with dense canopy; no signs of logging roads.	399	Proctor *et al.* (1983), in Malaysia
	306	Palm *et al.* (1999), Tropical rainforests
	300	World Agroforestry Centre (2011), Southeast Asia
	252	Prasetyo *et al.* (2000), Indonesia
	250	Houghton (1999); DeFries *et al.* (2002), the tropics
	230	Rahayu *et al.* (2005), Nunukan , East Kalimantan, Indonesia
	229	Omar (2010), Malaysia
	225	IPCC (2006), tropical Asia
	202	Hoshizaki *et al.* (2004), Primary dipterocarp forest in Pasoh Forest reserve, Peninsular Malaysia
	195	BAPPENAS (2010), Indonesia
	180	Laumonier *et al.* (2010); Southern Sumatra, Indonesia, disturbed and undisturbed forests
	177	Morel *et al.* (2011), Sabah, Malaysia
	164	Gibbs *et al.* (2007), for tropical Asia
	150	IPCC (2006) general data for tropical rainforest
	121	Griscom *et al.* (2009), pre-logged forest, Indonesia.
	55	Bryan *et al.* (2010), pre-logged forest, Papua New Guinea
	104	Stanley (2009), pre-logged forest, Indonesian territory of Papua
	93	Harja *et al.* (2011), Indonesia
	83	Pinard & Putz (1996), pre-logged forest, Malaysia
	61	Fox *et al.* (2010), pre-logged forest, Papua New Guinea
Average	189±87	

Table 1. Above ground carbon stocks (AGC) of different land use classes (continued).

Land use type and description [1]	AGC ($Mg\ ha^{-1}$)	Reference; remarks
DISTURBED UPLAND FOREST Natural forest area with logging roads and forest clearings.	250	World Agroforestry Centre (2011), logged forest, high density, Indonesia
	203	Rahayu *et al.* (2005), Nunukan , East Kalimantan, Indonesia
	180	IPCC (2006), for tropical Asia
	170	MoF (2008), Indonesia
	153	Saatchi *et al.* (2011) average of 43 M ha PNG forests with 30% canopy cover threshold
	150	World Agroforestry Centre (2011), logged forest, low density
	134	Omar *et al.* (2010), Malaysia
	132	Morel *et al.* (2011), average of 1970-2007 logged forest in Sabah, Malaysia.
	93	Palm *et al* (1999), logged forest, the tropics
	91	Griscom *et al.* (2009), above ground C pre-logging minus C lost from logging, the tropics
	87	Henson (2005a, 2009), logged forest, Malaysia
	74	Harja *et al.* (2011), Indonesia
	71	Stanley (2009), logged forest, PNG
	65	Morel *et al.* (2011), early secondary forest, Sabah, Malaysia
	60	Pinard & Putz (1996), logged forest, Malaysia
	57	Morel *et al.* (2011), medium disturbance secondary forest, Sabah, Malaysia
	55	Morel *et al.* (2011), late secondary forest, Sabah, Malaysia
	45	Fox *et al.* (2010), logged over forest, PNG
	43	Pinard & Putz (1996), logged over forest, Malaysia
	40	Bryan *et al.* (2010), logged over forest, PNG
	37	Bryan *et al.* (2010), logged over forest, PNG
Average	104±59	
UNDISTURBED SWAMP FOREST Forest wetland with temporary or permanent inundation	200	World Agroforestry Centre (2011), undisturbed swamp forest, Indonesia
	196	MoF (2008), Indonesia
	90	Harja *et al.* (2011), Indonesia
Average	162±51	
DISTURBED SWAMP FOREST Swamp forest with signs of logging canals, or degradation.	155	MoF (2008), Indonesian Forest Carbon Alliance study, Indonesia
	120	World Agroforestry Centre (2011), logged swamp forest, Indonesia
	78	Harja *et al.* (2011), Indonesia
	64	Morel *et al.* (2011), Sabah, Malaysia, low disturbance forest
	52	Morel *et al.* (2011), Sabah, Malaysia, high disturbance peat forest
	33	Morel *et al.* (2011), Sabah, Malaysia, medium disturbance swamp forest
Average	84±42	

Table 1. Above ground carbon stocks (AGC) of different land use classes (continued).

Land use type and description [(1)]	AGC (Mg ha^{-1})	Reference; remarks
UNDISTURBED MANGROVE Area along the coastline with high density of mangrove trees.	200	World Agroforestry Centre (2011), Indonesia
	170	Komiyama *et al.* (2008), Indonesia
	135	Putz & Chan (1986), study in Malaysia
	85	Harja *et al.* (2011), Indonesia
Average	148±43	
DISTURBED MANGROVE Logged-over and partly degraded mangrove area.	120	Komiyama *et al.* (2008), Indonesia
	105	Ong *et al.* (1982), Malaysia
	100	World Agroforestry Centre (2011), logged mangrove forest, Indonesia
	77	Harja *et al.* (2011), Indonesia
Average	101±15	
RUBBER PLANTATION Including rotational agroforestry rubber	97	Lasco & Pulhin (2004), rubber monoculture, Southeast Asia
	89	Palm *et al.* (1999), permanent agroforestry (jungle) rubber, the tropics
	46	Palm *et al.* (1999), rotational agroforestry (jungle) rubber the tropics
	53	Corpuzm *et al.,* (2011), monoculture, Philippines
	36	Prasetyo *et al.,* (2000), (jungle) rubber, Jambi, Indonesia
	31	World Agroforestry Centre (2011), estate on peat, Indonesia
Average	58±28	
OIL PALM PLANTATIONS Large-scale plantations recognizable in satellite images	60	Rogi (2002), Indonesia
	47	Syahrinudin (2005), recalculated based on biomass curve, Indonesia
	47	World Agroforestry Centre (2011), various kinds of estate, mainly rubber and oil palm
	40	van Noordwijk *et al*. (2010), averaged over 25 years, based on observations in Sumatra and Kalimantan, Indonesia
	40	Henson (2005b), estimated using OPRODSIM based on medium sized fronds, including oil palm roots and shoot, ground cover, pruned frond piles, shed frond base piles and male inflorescence piles, national average over 30 year
	36	Henson (2009), Malaysian national average over 30 year including the palm components as in Henson (2005b)
	31	World Agroforestry Centre, (2011), estate on peat (mainly oil palm), Indonesia
	30	Germer & Sauerborn (2008), the tropics
	29	Recalculated from Henson & Dolmat (2003) from a study of 1 to16 year old oil palm on peat in Malaysia: trunk (16 Mg C ha^{-1}), fronds (5.6 Mg C ha^{-1}), and male inflorescence (7.5 Mg C ha^{-1}) for a planting density of 160 palms ha^{-1}.
	26	Morel *et al.* (2011), Sabah, Malaysia
	23	Kheong (MPOC, unpublished), 45.3 t C ha^{-1} at 20 years after planting is considered to be the peak C stock; time-average C stock calculated as half of the peak C stock, Malaysia.
	23	Corley & Tinker (2003), Malaysia
Average	36±11	

Table 1. Above ground carbon stocks (AGC) of different land use classes (continued).

Land use type and description [(1)]	AGC ($Mg\ ha^{-1}$)	Reference; remarks
TIMBER PLANTATION Monoculture timber plantations	70	World Agroforestry Centre (2011), timber plantation, Indonesia
	60	World Agroforestry Centre (2011), timber plantation, Indonesia
	40	Matsumura *et al.* (2008), a study in Java of a 10-yr Acacia cycle interpolated from an 8-yr cycle, the most common cycle currently used.
	37.5	Nurwahyudi & Tarigan (2001) for *Acacia* 7 yr old, Indonesia
	37	Palm *et al.* (1999), for pulp trees in the tropics
	35	Matsumura *et al.* (2008), Peninsular Malaysia
	29	Morel *et al.* (2011), Sabah, Malaysia
Average	44±14	
MIXED TREE CROPS Also known as agroforestry.	77	World Agroforestry Centre (2011), agroforest on peat, Indonesia
	30	Rahayu *et al.* (2005), Nunukan , East Kalimantan, Indonesia
Average	54±24	
UPLAND SHRUB LAND Upland (well drained sols), small trees and shrubs	35	IPCC (2006) for tropical shrub land
	30	Istomo *et al.* (2006), Indonesia
	29	Jepsen (2006), Sarawak, Malaysia
	27	World Agroforestry Centre (2011), Indonesia
Average	30±3	
SWAMP SHRUB LAND Wetland (periodically or permanently inundated), small trees and shrubs	35	IPCC (2006) for tropical shrub land
	30	Istomo *et al.* (2006), Indonesia
	29	Jepsen (2006), Sarawak, Malaysia
	18	World Agroforestry Centre (2011), shrub on peat, Indonesia
Average	28±6	
INTENSIVE AGRICULTURE Open area, usually intensively managed for annual row crops.	12.5	Hashimotio *et al.*, (2000) based on biomass estimates of 50 $Mg\ ha^{-1}$ for 10-12 yr fallow rotation in Kalimantan, Indonesia
	12	World Agroforestry Centre (2011), cropland, Indonesia
	10	Murdiyarso & Wasrin (1996), Indonesia
	8	World Agroforestry Centre (2011), cropland on peat, Indonesia
Average	11±2	
SETTLEMENTS Homestead, urban, rural, harbor, airports, industrial areas.	10	BAPPENAS (2010), assuming one third of the homestead area is allocated for home gardens (mixed tree crops and agriculture), Indonesia
	4	World Agroforestry Centre (2011), Indonesia
Average	7±3	
GRASSLAND Upland (well drained soils), dominated by grasses.	4	Rahayu *et al.* (2005), Nunukan, East Kalimantan, Indonesia
	2	World Agroforestry Centre (2011), time-averaged value, Indonesia
Average	3±1	

Table 1. Above ground carbon stocks (AGC) of different land use classes (continued).

Land use type and description [1]	AGC ($Mg\ ha^{-1}$)	Reference; remarks
SWAMP GRASSLAND Wetland (periodically or permanently inundated) dominated by grasses	2	Palm *et al.* (1999), the tropics
RICE FIELD Paddy field usually irrigated.	2	Palm *et al.* (1999), the tropics
COASTAL FISH POND Open area on coast always inundated	0	Assumed
BARE SOIL Area with little or no woody vegetation	36	Recommended as a default value when modeling CO_2 emissions from land use change linked to oil palm, because it is a transitional category with various original land cover source [2]
MINING Open area with mining activities.	0	Assumed

[1] The detailed description is provided by Gunarso *et al.* (2013, this publication).
[2] Assumed to be the same as that of oil palm plantation. The C stock is mostly in the form of necromass.

Carbon Stock in Mineral Soils

Globally, soils store about 3.3 times the amount of C present in the atmosphere and about 4.5 times the C found in above ground terrestrial biota. The soil carbon stock varies with land use and land management systems; hence, the uncertainty in soil carbon stock data is high. Despite the advances in soil survey around the world, data on soil bulk density is scarce relative to that on soil organic carbon content. Both variables are needed for the calculation of volume-based soil organic C stock and its possible change; consequently, a modeling approach is required to fill the gap between the available soil data in order to produce a soil carbon assessment.

Carbon stock in the top 30 cm of soil in humid tropical forests ranges from 5 to 180 $Mg\ ha^{-1}$ (IPCC, 2006) and changes in soil carbon content are influenced by various factors such as soil tillage and organic matter inputs. Mean estimates of carbon stock for humid tropical soils suitable for oil palm may be as high as 120 ± 60 $Mg\ C\ ha^{-1}$ (Germer & Sauerborn, 2008) and as much as 30% of soil organic matter may be lost when forest is converted to plantations (Murty *et al.*, 2002). This would translate into an initial carbon loss of about 36 ± 18 $Mg\ C\ ha^{-1}$ when the land is converted to a plantation, but when low biomass land cover types are converted to plantations, soil carbon stock might increase. However, there are many inconsistencies and uncertainties associated with soil carbon stock change as affected by land use change in mineral soils, especially from land use change from forest to oil palm plantations (Table 2). Most problematic is the fact that data for initial carbon stock are generally not available. Consequently, it is not possible to make reliable conclusions regarding the dimensions of CO_2 emissions from mineral soil carbon, and hence this component of CO_2 emissions is not considered in the analysis by Agus *et al.* (2013 – this publication).

Table 2. Reported change in carbon stock in mineral soil as affected by land use change

Initial land use	Subsequent land use	Change in C stock, references
Logged forest	Oil palm	32% and 15% increase of soil organic carbon in the 0-45 cm layer, in the first and second cycles respectively, of oil palm under intensive organic matter management Mathews *et al*. (2010).
Oil palm, 5 years after planting	Oil palm, 20 years after planting	Increase of soil organic carbon (C_{org}) in the avenue and weeded circles from 0.82% to 2.21%. Increase of C_{org} from 0.82% to 3.09% in the pruned frond windrows occupying 20% of the area and receiving an equivalent of 4.8 Mg C ha^{-1} yr^{-1} from palm fronds. (Haron *et al*. 1998).
Primary forest	Secondary forest, and oil palm plantations	C_{org} was 29±9 g kg^{-1} and 21±8 g kg^{-1} under the canopy and gap areas respectively of a primary forest, 17±3 g kg^{-1} and 14±4 g kg^{-1} under the canopy and gap area of a secondary forest and 16 ±8 g kg^{-1} under an oil palm plantation. The three land cover types were adjacent to each other in Pasoh, Peninsular Malaysia (Adachi *et al*. 2006).
Secondary forest, 30 years after logging	Oil palm 9 and 19 years old, rubber 30 years old	No significant change from about 33 g kg^{-1} in 0 – 10 cm soil depth (Tanaka *et al*. 2009).
Forest	Long term agricultural cultivation	30% decrease in soil C stock (Murty *et al*. 2002) in soils suitable for oil palm with 120±60 Mg C/ha (IPCC, 2006)
Forest	Degraded land	50% decrease in soil C stock (Murty *et al*., 2002; Germer & Sauerborn, 2008).
Forest	No tillage system	Increase of 0-10% organic C with crop residue recycling (Murty *et al*., 2002; Germer & Sauerborn, 2008).
Forest	Plantation	30% decrease in soil C stock (Murty *et al*. 2002; Germer & Sauerborn, 2008).
Degraded land	Plantation	30% increase in soil C stock (Murty *et al*., 2002).

EMISSIONS FROM PEAT SOILS

Distribution and Carbon Stock of Peat Soil

Peat soil is one of the most important sites for carbon storage under tropical forest conditions. Carbon is stored in plant biomass above and below ground, in necromass and in the soil, the largest stock of carbon in peat soil being in the below-ground peat itself. For example, a one meter layer of peat stores between 300-700 Mg C ha^{-1} (Page *et al*., 2002; Agus & Subiksa, 2008); in contrast, the above ground biomass of a primary forest stores only 90-200 Mg C ha^{-1} (Table 1). The carbon rich organic matter in peat builds up under the anoxic conditions characteristic of swamp forests over 3000 to >8000 years. Once the forest is cleared and drained, however, peat will be decomposed by oxidization and a peat formation can disappear within decades (Parish *et al*., 2007; Hooijer *et al*., 2006; Rieley & Page, 2008). The wide-scale conversion of peat formations and the resultant oxidation of peat soils represent a very large source of actual and potential CO_2 emissions.

The earlier estimate of Indonesian peat soil area was about 21 Mha (Wahyunto *et al*., 2004, 2005, 2006), which is equivalent to about 83% of the reported peat soil of Southeast Asia and which stores an estimated 37.2 Pg of carbon (Hooijer *et al*., 2006; Wahyunto *et al*., 2004, 2005, 2006). However, these estimates were based on maps generated using Landsat TM images with little ground truth data, especially for Papua. Soil surveys have progressed in Indonesia and field data have been plotted against an alternative map of peat soils to produce a revised estimate of Indonesian peat soil area of 14.9 Mha (Ritung *et al*., 2011). The greatest reduction in area was in Papua where soil survey data were poor and the estimated extent of peat was reduced by more than 50% (Table 3). The extent of peat soils in Sumatra and Kalimantan each showed a reduction of around one million hectares, estimates that are in line with other recently published values of 13.0 Mha (Miettinen *et al*., 2012).

A study of two peat domes in South Sumatra (Airsugihan and Telukpulai), three in Central Kalimantan (Sebangau, Block B and Block C) and one in West Papua (Teminabuan) used a 3D modeling approach using optical images from Landsat ETM+ and synthetic aperture radar data from the NASA Shuttle Radar Topographic Mission (Jeanicke *et al*., 2008). The sites in Central Kalimantan and South Sumatra were selected because of their representative character and the availability of around 750 peat thickness measurements; Teminabuan was chosen to extend the geographical range of the study and to include another type of Indonesian peat dome in the modeling process, even though detailed peat thickness data were lacking for that locality. The results from this five dome study were then extrapolated across the nation based on three key assumptions: average peat depth of 4.5±0.85 m, total peat soil area of 21 Mha as projected by Wetlands International (Wahyunto *et al*., 2004; 2005; 2006), and average carbon content of 58 kg m^{-3}. The total carbon store in Indonesian peat formations was then estimated to be 55±10 Pg (Jaenicke *et al*., 2008).

Subsequently, field based verification of the Wetlands International peat soil maps led to a revised and stratified peat soils map with 5.2 Mha of shallow peat (50-100 cm), 3.9 Mha of medium deep peat (100-200 cm), 2.9 Mha of deep peat (200-300 cm) and 3.0 Mha of very deep peat (>300 cm), giving a total of 15 Mha (Ritung *et al*., 2011). The very deep peat may reach beyond 800 cm at the center of some domes, but the overall average peat thickness is unlikely to exceed 300 cm (Ritung *et al*., 2011), although some authors estimate mean thickness at between 550 and 700 cm (Miettinen *et al*., 2012). If one assumes 300 cm is the average peat depth and 60 kg C m^{-3} the average carbon content (Page *et al*., 2002), then the estimated carbon storage for the 15 Mha of Indonesian peat formations would be approximately 27 Pg (1800 Mg C ha^{-1}), about one half the 46.6 Pg C estimated by Page *et al*. (2011b) and almost a third of the 55±10 Pg estimated by Jeanicke *et al*. (2008).

In Malaysia, a recent estimate of peat soil area is 2.4 Mha (Table 4), with about two thirds of the total being found in Sarawak; estimates of the carbon stored in Malaysian peat soil ranges from 7.9 to 9.2 Pg (Page *et al*., 2011a). In Papua New Guinea, the distribution and extent of peat soil is not well documented, ranging from 0.05 to 2.9 Mha, with the best estimate around 1.1 Mha and peat carbon stock estimated at about 1.4 Pg, and ranging between 0.6 to 1.7 Pg (Page *et al.*, 2011b).

Table 3. Areas (Mha) of peatland in Sumatra, Kalimantan and Papua, Indonesia as reported by three sources.

Region	Wahyunto *et al.* (2003, 2004, 2006)	Ritung *et al.* (2011)	Miettinen *et al.* (2012)
Sumatra	7.2	6.4	7.2
Kalimantan	5.8	4.8	5.8
Papua	7.8	3.7	n.a.
Total	20.8	14.9	>12.0

n.a. : Not available

Table 4. Extent of peat soils for the three regions of Malaysia as reported by three sources

Region	Gunarso *et al.* (2013)	Omar *et al.* (2010)	Miettinen *et al.* (2012)
Peninsular	719,909	716,944	854,884
Sarawak	1,308,086	1,588,142	1,442,845
Sabah	117,035	121,514	191,330
Total	2,145,030	2,426,600	2,489,059

Greenhouse Gas Emission Due to Peat Oxidation

Land use change from peat forest to plantation, especially for those plantations requiring relatively deep drainage, will change the function of the peat soil from a net carbon sequester to a net carbon emitter (Parish *et al*., 2007; Agus & Subiksa, 2008). Numerous studies have shown that peat oxidation due to drainage is a long-term process that will create a long-term source of CO_2 emissions (Stephen & Johnson, 1951; Stephen, 1956; Wösten *et al*., 1997). Data on the dimensions of these emissions vary widely as there are many interacting factors influencing this process. The most frequently reported factor determining CO_2 emission from peat is the depth of the groundwater table, which is affected by drainage (Hooijer *et al*., 2010, 2012; Couwenberg *et al*., 2010; Jauhiainen *et al*., 2005, 2012; Page *et al*., 2011a; Husnain *et al*., Pers. Comm.; Dariah *et al.* Pers.Comm.). The stored carbon may be lost from biomass, necromass and peat soil by burning and/or decomposition, and deep drainage (i.e., greater than 60 cm) greatly increases the rate of peat oxidation and the risk of peat fire (Page *et al*., 2002; van der Werf *et al*., 2008).

In addition to CO_2, methane (CH_4) and nitrous oxide (N_2O) are also emitted during land conversion particularly during fire events; nonetheless, CO_2 dominates the GHG emission profiles linked to land use on peat soils due to the total volumes of CO_2 emitted, even though CH_4 and N_2O have greater global warming potentials (GWPs): 21 for CH_4 and 296 for N_2O in comparison with CO_2 (IPCC, 2006). For example, CH_4 emissions occur under anaerobic conditions through the action of methanogenic bacteria (Holzapfel-Pschorn & Conrad, 1985), but when the water table is deeper than 20 cm CH_4 emissions are rarely detectable. The availability of easily decomposable material such as leaf litter, which is abundant on the surface in relatively undisturbed sites, is an important factor promoting CH_4 emission (Jauhainen *et al.,* 2008). These CH_4 fluxes in undrained forest represent only about 0.9% of GHG emission in the form of CO_2-e (Jauhiainen *et al.,* 2005; Inubushi *et al.,* 2003), while in drained forests and agricultural areas CH_4 emission levels represent only 0.01% to 0.2% relative to that of CO_2 (Melling *et al.,* 2005; Jauhiainen *et al.,* 2008).

Similarly, N_2O is emitted as a by-product of nitrification (conversion of NH_4^+ to NO_3^-) and denitrification (conversion of NO_3^- to N_2O or N_2) under low O_2 availability (Inubushi *et al.,* 2003). Increased availability of NO_3^- enhances N_2O emissions from soils (Yanai *et al.,* 2007) and the relative contribution of N_2O released from agricultural land can be very high. Nonetheless, the range of measured N_2O emission varies widely depending on many factors linked to management practices and transient weather events; thus, any modeled estimate of GHG emission based on regional or landscape level assumptions are inherently uncertain. Consequently, N_2O emissions were not considered as part of a regional effort to estimate GHG emissions linked to palm oil production (see Agus *et al*., 2013 – this publication). It should be noted, however, that by only focusing on CO_2, the total GHG emissions will be somewhat underestimated.

In some instances CO_2 emission from the surface of peat forest can be higher than that from peat under oil palm, which can be attributed to the contribution of CO_2 by root-related respiration that is higher under forest due to higher root density and activity (Melling *et al*., 2005). However, this increased emission represents recycled CO_2 fixed by photosynthesis and thus does not represent a net increase in atmospheric CO_2. In the rhizosphere, a term used to describe the soil zone dominated by the roots, bacterial and fungal respiration is dependent on inputs from the living roots and, although it is not 'autotrophic' in the original meaning of the term, many researchers who study peat refer to all respiration linked to current and recent photosynthesis as being 'autotrophic'. The proportion of plant-based respiration (e.g., autotrophic) to peat-based respiration (heterotrophic) is presently a source of uncertainty. Two approaches can be taken to address this problem.

(i) Separation of plant-based from peat-based respiration by the use of root exclusion or isotope labeling techniques and;
(ii) Monitoring carbon stock change (bulk density and carbon content changes with peat depth) of different land use/land cover types.

Without consistent use of such approaches there will continue to be uncertainty concerning the precise effects of agricultural operations and oil palm expansion on peat CO_2 emission.

Research in temperate zones has found that 55-65% of peat respiration was generated via root+rhizosphere interactions, which are considered to be autotrophic, and that only about 35-45% of the soil respiration could be classified as a GHG emission due to the decomposition of peat (Knorr *et al*., 2008). In another study, the contribution of peat-related decomposition was shown to be as high as 42%, while root+rhizosphere respiration was 41% and the remainder, 17%, was the consequence of above ground litter decomposition (Mäkiranta *et al*., 2008). Root-related respiration in oil palm plantations in Southeast Asia has been found to be 38% and 40% of the total measured at the soil surface by closed chambers (Agus *et al*., 2010; Melling *et al*., 2007). In transects established in *Acacia* plantations in Riau province, Indonesia, CO_2 emission near the trees was about 21% higher than at the midpoint between trees, a difference that was attributed to autotrophic respiration linked to roots (Jauhiainen *et al*., 2012). Unlike oil palm plantations, however, planting density in *Acacia* plantations is high (2 m x 2 m) and all areas in these plantations are probably influenced by roots. The root-related autotrophic component of different land cover types is therefore uncertain, and adopting total CO_2 efflux data will overestimate net CO_2 emissions.

For oil palm plantations on peat, published reports from closed chamber measurements of soil surface flux range from 20 to 57 Mg CO_2 ha^{-1} yr^{-1}, with an average value of 38 Mg CO_2 ha^{-1} yr^{-1} (Reijnders & Huijbregts, 2008; Wicke *et al.,* 2011; Murdiyarso *et al.,* 2010; Murayama & Bakar, 1996; Jauhiainen *et al.,* 2001; Melling *et al.,* 2005; Melling *et al.,* 2007; Agus *et al.,* 2010). Recent studies in Jambi, Sumatra fall within the middle of this range, with mean values corrected to discount for plant-based, or autotrophic, respiration of 38±2 Mg CO_2 ha^{-1} yr^{-1} for 6 year old oil palm and 34±3 Mg CO_2 ha^{-1} yr^{-1} for 15 year old oil palm (Dariah *et al.,* Pers. Com.). Similarly, new studies from Sumatra and Kalimantan found CO_2 emissions under oil palm plantations on peat varied widely from 18±13 to 66±24 Mg CO_2 ha^{-1} yr^{-1} with the overall average of 39±19 Mg ha^{-1} yr^{-1}; the highest CO_2 emission was observed in oil palm plantations in Riau (Husnain *et al.,* Pers. Com.).

Another approach for estimating CO_2 emissions from peat soils is based on measurements of subsidence over time, which when coupled with the monitoring of changes in bulk density and carbon content, can provide an independent estimate of peat oxidation. Recent studies in Riau and Jambi Provinces of Indonesia exemplify the subsidence technique and provide a different, and much larger, estimate of net CO_2 emissions (Hooijer *et al.,* 2012). However, the experimental design of this study did not account for potential differences in bulk density within the soil profile and the initial mean bulk density of the soil was assumed to be the same as the bulk density measured just below the average water table depth of the subsequent land use. In addition, the model used to estimate changes in carbon stock assumed a constant carbon content of 55% throughout the soil profile and across all sites—an assumption that disregards spatial variability and changes in carbon content linked to the degradation of peat over time. Carbon content of peat is variable and is the basis of the peat classification system which defines "fibric"," hemic" and "sapric" types of peat; essentially, as peat is oxidized, it becomes more carbon dense (Wurst *et al.,* 2003). In summary, the study by Hooijer *et al.* (2012) estimated soil decomposition to represent about 92% of subsidence and the remaining 8% was attributed to shrinkage and compaction, which produced a modeled emission estimate of 100 CO_2 Mg ha^{-1} yr^{-1} for the first 25 year cycle of an oil palm plantation operating on peat soils, or a value of 95 Mg CO_2 ha^{-1} yr^{-1} when annualized over a 30-year rotation cycle (Page *et al.,* 2011a).

Other studies have shown that the decomposition component of land subsidence was about 60% (Wösten *et al.,* 1997), 60% (Hooijer *et al*., 2010) or 40% (Couwenberg *et al*., 2010). In the Everglades region of Florida, long-term studies of peat subsidence following conversion to agriculture have shown losses of about 40% of their original volume in the 40 years since the onset of drainage (Stephen & Johnson, 1951). Although these studies unequivocally document that peat oxidation following drainage is a long-term source of CO_2 emissions, they have also demonstrated that the initial cause of subsidence after drainage is due to physical compaction (Stephen & Johnson, 1951; Stephen, 1956; Wösten *et al.,* 1997).

As stated previously, all of these estimates are contingent upon water table depth and Hooijer *et al*. (2006, 2010) developed a model that correlates drainage depth with CO_2 emissions such that for each 1 cm of drainage depth there is an emission of about 0.91 Mg CO_2 ha^{-1} yr^{-1}. For a typical oil palm plantation with a water table situated at about 60 cm below the soil surface, the estimated emission would be about 54 Mg CO_2 ha^{-1} yr^{-1}. However, this relationship is based largely on experiments using closed chambers in which there was no separation between autotrophic respiration mediated by roots and heterotrophic respiration linked to microbial decomposition (Hooijer *et al*., 2006). In order to avoid over estimating CO_2 emissions by using total soil respiration, we recommend using the emission factor developed by Hooijer *et al*. (2010) modified by a coefficient of 0.79 to correct for the root-related emission based on the studies of Jauhiainen *et al*. (2012). The complete equation is therefore:

$$E_{bo}\ (\mathrm{Mg\ CO_2\ ha^{-1}\ yr^{-1}}) = 0.91*0.79*\ \text{drainage depth (cm)} \quad [3]$$

Using Equation 3 for an oil palm plantation with a water table depth that varies between 50 and 70 cm gives estimated emissions that range between 36 to 50 Mg CO_2 ha^{-1} yr^{-1} with an average of 43 Mg CO_2 ha^{-1} yr^{-1}. This is the value we recommend as a default when estimating emissions from oil palm plantations operating on peat soils.

Emissions Due to Burning

Fires have direct on-site effects resulting in degradation of vegetation, loss of biodiversity, destruction of property and occasional loss of life, while off-site impacts include carbon emissions, smoke and its impacts on human health. Wild fire can be caused by natural phenomena such as lightning, but human activities, particularly land preparation for agriculture and plantation estates, are among the most important causes (FAO, 2011; Herawati & Santoso, 2011.).

The impacts of fire on GHG emissions in Southeast Asia are considered to be of historical significance and loom large in any discussion or estimate of CO_2 emission and land use. The largest single source of emission in recorded history is believed to be the GHG emissions from forest and peat fires in Southeast Asia during the exceptionally strong 1997/98 *El Niño* event, which led to the release of an estimated 2.9 to 9.4 Pg CO_2 (Page *et al.,* 2002). In the last decade, a combination of remote sensing data and top-down models have been used to monitor the annual variation in fire related emissions, which have fluctuated between 0.09 and 1.3 Pg CO_2 yr^{-1} (van der Werf *et al.,* 2008, 2010). Annual estimates are highly variable, and during the average 2006 *El Niño*, fire emission in Kalimantan was more than 30 times greater than those during the 2000 *La Niña*, which is an exceptionally wet episode that alternates with *El Niño* droughts (van der Werf *et al.,* 2010).

Estimates of the impacts of the depth of fire on peat soils are dominated by a limited number of studies that have focused on observations made during *El Niño* years in Central Kalimantan. These values range from approximately 50 cm in 1997 (Page *et. al*., 2002) to 39 cm in 2002 (Usup *et al.*, 2004) and 33 cm in 2006 (Balhorn *et al.,* 2009). These published values should be viewed with caution, because water table depth and the distribution of rainfall both influence the extent and intensity of fire. Nonetheless, fire has been used historically as a management tool when preparing land for new oil palm plantations, in spite of the legal proscriptions limiting its use (Someshwar *et al*., 2011). Unfortunately, precise information as to the intensity and depth of peat fires during average or wet years is not available, but evidence from remote sensing indicates, and our own field experience supports, the supposition that the depth of peat fires during average or wet years is only a fraction of the levels documented during *El Niño* droughts (van der Werf *et al.,* 2010).

Consequently, we recommend using relatively conservative values when estimating the impact of historical fire on peat soils during plantation establishment over decadal time periods that span both wet and dry years. Specifically, we assume that the average depth of a peat fire would be 15 cm for swamp forest and 5 cm for swamp shrub land (Agus *et al.* 2012); the difference between the two values is based on anecdotal accounts that greater levels of above ground biomass lead to more intense fires and deeper burns. Moreover, we assume there is no burning of peat during oil palm replanting or the conversion of other land uses that have already been cleared for agriculture, agroforestry or other forms of plantation agriculture. Calculation of our emission factors for peat fires is based on an average carbon density of 0.06 Mg m^{-3} for peat soils (Page *et al*., 2002), which translates into emissions factors of 330 and 110 Mg CO_2 ha^{-1} for swamp forest and swamp shrub land respectively. The derivations of these emissions factors are based solely on assumptions and logic, but we feel this is preferable to ignoring a significant source of emissions due to the lack of empirical data.

Assessment of Historical Emissions

Based on the discussion in the previous sections, Table 5 summarizes C stock in plant biomass, peat oxidation loss and related water table depths, and emissions from burning. Only emission from above ground biomass, peat soil organic matter oxidation and controlled peat fire were taken into account in our analysis (Agus *et al.* 2013 – this publication). For peat soil, there are more data based on instantaneous CO_2 efflux than calculated from carbon stock change, while for living biomass most data are based on carbon stocks. The emission factor, multiplied by the activity data will give the emission estimate for the land areas of interest. Equation [2] can be rewritten in term of CO_2-e emission as,

$$Total\ Emission = \sum_{ij} A_{ij}\ [Emission_{ij\ LB} + Emissionij\ SOIL]/Tij \quad [4]$$

Where

A_{ij} = the activity data or area of land use under land cover type *i* that changes to type *j*

$Emission_{ij\ LB}$ = change in carbon stock in the living biomass under land cover type *i* that changes to type *j* * 3.67 (to convert C to CO_2). A_{ij} is presented outside the diagonal of the land use change matrix. Land use that is unchanged appears in the diagonal of the land use change matrix and is assumed not to exchange CO_2 from the living biomass with that in the atmosphere. While this is not true in the short term, it holds in the long term (over one plantation cycle or longer). Deviation from this assumption may occur because of changes in land management.

$Emission_{ij\ SOIL}$ = change in peat carbon stock due to oxidation from drainage and burning under land cover type *i* that changes to type *j* * 3.67. For peat soil land uses that remain the same during the analysis period, drainage oxidation is calculated as A_{ii} * peat oxidation rate under that particular land use (in Mg CO_2 ha^{-1} yr^{-1}). Emission from drainage oxidation of peat soil that changes in land use from *i* to *j* = the average of emissions from the two land uses * A_{ij}.

T_{ij} = the time scale of calculation

In a separate paper (Agus *et al.*, 2013 – this publication), estimates of total CO_2 emissions from land use linked to the establishment and operations of oil palm plantations in Malaysia, Indonesia and Papua New Guinea has been carried out by combining land use change matrices that cover three consecutive periods (Gunarso *et al.,* 2013 – this publication) with the emission factors recommended by this paper (Table 5).

Table 5. Mean above ground carbon (AGC) stocks (see Table 1) used for the calculation of CO_2 emissions due to land use change (LUC); the water table depth and associated CO_2 emission factors for peat oxidation and the CO_2 emission factors from peat burning in Indonesia, Malaysia and Papua New Guinea.

Land use/land cover type	AGC (Mg ha^{-1})	Water table depth (cm)	CO_2 emissions from peat oxidation (Mg ha^{-1}yr^{-1})	CO_2 emissions from fire on peat due to land use change (Mg ha^{-1})
Undisturbed Forest	189			
Disturbed Forest	104			
Undisturbed Swamp Forest	162			330
Disturbed Swamp Forest	84	30	22	330
Undisturbed Mangrove	148			
Disturbed Mangrove	101			
Traditional Rubber Plantation	55	50	36	
Oil Palm Plantation	36	60	43	
Timber Plantation	44	50	36	
Mixed Tree Crops	54	50	36	
Shrub land	30			
Swamp Shrub land	28	30	22	110

Table 5. Mean above ground carbon (AGC) stocks (continued)

Land use/land cover type	AGC ($Mg\ ha^{-1}$)	Water table depth (cm)	CO_2 emissions from peat oxidation ($Mg\ ha^{-1} yr^{-1}$)	CO_2 emissions from fire on peat due to land use change ($Mg\ ha^{-1}$)
Annual Upland Crops	11	30	22	
Settlements	7	70	50	
Grassland	3			
Swamp Grassland	2	30	22	
Rice Field	2	10	7	
Coastal Fish Pond	0			
Bare soils	36[1]			
Mining	0	100	72	
Water Bodies	0			
No Classification	0			

[1]Bare soils is a transitional category of unknown precedence and the value of 36 $Mg\ ha^{-1}$ is recommended in order to avoid introducing artifacts into the estimation of net oil palm emissions

CONCLUSIONS

This report reviews the scientific literature on carbon stocks for different land cover types in Southeast Asia; these values can be used to calculate CO_2 emission factors due to land use change (see Agus *et al*, 2013 – this publication). In addition, we provide a review of the dimensions of the recurrent CO_2 emissions due to the oxidation of peat following drainage and provide a framework for estimating the one-time emissions caused by peat fires at the time of plantation establishment (see Table 5). There is a high degree of variation in all of these sources of emission which will contribute to uncertainties in any CO_2 emission analysis.

The reported values for plant biomass carbon stock reflect the inherent variation in natural habitats and disturbance intensities caused by human intervention. The recommended values for calculating emission factors from land use change between any two land cover categories are the differences between the mean carbon stock values for the two categories (Tables 1 and 5). In the case of natural or quasi-natural land cover types, these are not time-averaged values, but are assumed to reflect the carbon stocks at the time of conversion. This is done to avoid confounding CO_2 emissions from degradation due to logging and wildfire with the emissions specifically due to the clearing of land for agriculture. In contrast, the carbon stock values for human modified land cover types are the time averaged values that reflect the cyclical harvest or renovation period characteristic of each production system, which in the case of oil palm is based on the 25 year cycle typical for oil palm plantations.

The source of the uncertainty in the estimates of CO_2 emissions linked to the oxidation of peat is largely the consequence of the methodological limitations of the two major approaches for measuring (closed chamber systems) or modeling (tracking subsidence) the decomposition of peat following drainage. The values produced by the two methodological approaches vary widely and the emission factor recommended as a default value (43 $Mg\ CO_2\ ha^{-1}yr^{-1}$) is based on our evaluation of the various published studies and the assumption that water tables in oil palm plantations are at approximately 60 cm from the soil surface. Unlike the emissions factors from land use change and peat fires, which are one-time events, the emissions from the oxidation of peat recur annually until the active drainage of the land cover type is ended. This is not only true for human managed land cover types, such as oil palm and tree plantations, but also for disturbed swamp forests and shrub lands that have been impacted by logging canals.

The emission factors reported for peat fires are also uncertain, due to the lack of published studies that document the phenomenon, compounded by the variation in fire intensity linked to inter-annual climate variability. Peat fires burn deeper in drought years but occur only superficially or are absent during wet years. We provide emissions factors only for peat fires linked to the conversion of swamp forest and shrub land to oil palm plantation and these values are based on anecdotal evidence that the use of fire to clear biomass has been a standard operating procedure over the last two decades (Table 5). No emissions factors are provided for peat fires that impact other land cover categories or other types of land use change.

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OIL PALM AND LAND USE CHANGE IN INDONESIA, MALAYSIA AND PAPUA NEW GUINEA

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ABSTRACT

Land use change associated with the expansion of industrial scale oil palm plantations in three regions of Indonesia (Sumatra, Kalimantan, and Papua), in Malaysia, and in Papua New Guinea, was documented using Landsat images that were visually interpreted to create a region-wide map of 22 different land cover types spanning three temporal periods (1990 to 2000, 2001 to 2005 and 2006 to 2009/2010). In 1990, there were approximately 3.5 Mha of industrial oil palm plantations in the three countries, which had expanded to 13.1 Mha hectares by 2010. Growth occurred at an approximately constant rate of 7% per year over twenty years; the absolute rate of expansion was greatest in Sumatra in the first and second period (167,000 and 219,000 ha yr^{-1}), which was surpassed in Kalimantan in the last temporal period (360,197 ha yr^{-1}). When averaged over all regions and temporal periods only 4.1% (397,000 ha) of oil palm plantations originated on land derived directly from undisturbed forests (0.2% upland and 4.0% swamp), while 32.4% (3.1 Mha) were established on land previously covered with disturbed forest (25.6% upland and 6.8% swamp). Conversion of low biomass shrub lands and grasslands was documented at 17.8% (1.7 Mha) with 13.5% from upland soils and 4.4% from swamp soils; plantations and agroforest combined contributed 33.9% (3.3 Mha). A category recognized as bare soil, the result of change involving multiple different classes, including the replanting of mature oil palm plantations and the conversion of forest, represented 8.3% (0.8 Mha); miscellaneous categories including annual crops, mines, settlements, mangrove swamps, water bodies, and persistent clouds totaled 3.4% (334,000 ha).

Forest conversion to establish oil palm, including both undisturbed and disturbed forest in both upland and swamp forest habitats summed over all temporal periods was proportionally greatest in Papua (61%: 33,600 ha), Sabah (62%: 714,000 ha) and Papua New Guinea (54%: 41,700 ha), followed by Kalimantan (44%: 1.23 Mha), Sarawak (48%: 471,000 ha), Sumatra (25%: 883,000 ha) and Peninsular Malaysia (28%: 318,000 ha). In Kalimantan, the largest sources of land for new plantations were actually from shrub and grassland (48%: 1.3 Mha), while other types of plantations were more important in Sumatra (59%: 2.1 Mha) and Peninsular Malaysia (44%: 487,000 ha). In Indonesia, the largest single cause of historical forest loss can be attributed to unsustainable logging followed by the impact of fire, which in combination led to the progressive transition of large areas of forest landscape into agroforest or shrub land. In Malaysia, the direct conversion of forest to oil palm was more common, particularly in Sabah and Sarawak, but in Peninsular Malaysia the conversion of other types of land use; particularly plantation crops such as rubber, were more important.

A separate analysis using an existing data set for peat soils showed oil palm plantations on peat increased from 418,000 ha (12% of total oil palm area) in 1990 to 2.43 Mha (18%) by 2010 for the total study area. Sumatra has the largest absolute extent of oil palm plantations on peat (1.4 Mha: 29%), followed by Sarawak (476,000 ha: 46%), Kalimantan (307,515 ha: 11%), and Peninsular Malaysia (215,984 ha: 8%), with only 2% of oil palm plantations occurring on peat in Sabah (29,000 ha) and Papua (1,727 ha), while there was no conversion of peat soils in Papua New Guinea.

Key words: *Southeast Asia, land cover, land use change, deforestation, oil palm, peat, forest.*

INTRODUCTION

Land use and land cover change in palm oil producing countries is cited as one of the main drivers of deforestation, particularly in Indonesia and Malaysia which produce approximately 85% of the world's palm oil (USDA-FAS, 2010). The absolute rate of deforestation in Indonesia is considered to be among the highest on the planet, and has been estimated to fluctuate between 0.7 and 1.7 Mha yr^{-1} between 1990 and 2005 (Hansen *et al.,* 2009). Malaysia lost approximately 230,000 ha of forest habitat annually between 2000 and 2010, but since the total forest area is less, the proportional rate is actually higher when compared to Indonesia, averaging almost 1.4% of the total forest area annually in the last decade (Miettinen *et al.,* 2012a). During this period, the palm oil sector grew dramatically in both countries, expanding from less than 3.5 Mha in 1990 to more than 9.5 Mha in 2005 (Teoh, 2009), and numerous reports in both the scientific and popular media have linked the expansion of oil palm plantations with deforestation.

Originally, the issue of deforestation and oil palm focused on the negative impacts on biodiversity and traditional communities (Fitzherbert *et al.,* 2008; Marti 2008; Sheil *et al.,* 2009), but the discussion soon expanded to cover climate change as palm oil was used increasingly as a feedstock for biofuels (Germer & Sauerborn, 2008; Gibbs *et al.,* 2008). The expansion of oil palm is responsible for emissions of greenhouse gases (GHG) when new plantations replace forest habitat because the amount of carbon stored in their stems, leaves and roots is small compared with the carbon stocks of the natural forests they replace (Wicke *et al.,* 2008). In addition, the expansion of the palm oil sector is linked to the drainage and conversion of peat soils, which creates two additional sources of CO_2 emissions: A one-time emission due to soil fire and recurrent annual emissions due to soil drainage. Although currently illegal in both Malaysia and Indonesia, fire has been used historically to clear vegetation at the time of plantation establishment (Someshwar *et al.,* 2011). If the top layers of the peat are dry, these will catch fire and burn down into the soil profile until the peat is sufficiently humid to extinguish the fire. Subsequently, the upper horizons of the peat soil profile are drained to create the conditions necessary for oil palm cultivation; this changes the ecological processes of the soil biota and leads to the gradual oxidation and decomposition of the peat matrix and, as a consequence, the release of CO_2 (Agus *et al.,* 2009; Hoojer *et al.,* 2006, 2010).

Numerous recent studies have addressed deforestation in the region, but either they have not directly addressed the specific issues of oil palm plantations (Stibig & Malingreau, 2003; Miettinen *et al.,* 2012a, Hansen *et al.,* 2009, Broich *et al,.* 2011, Ekadinata & Dewi, 2011; Margono *et al.,* 2012), or have been conducted at a scale that does not allow for a comprehensive evaluation of the oil palm sector as a whole (SarVision, 2011; Carlson *et al.,* 2012a, 2012b; Miettinen *et al.,* 2012b, 2012c). Perhaps more importantly, these studies have not adequately documented the full range of land cover types that are converted to oil palm, nor evaluated land use change linked to the palm oil sector in the context of other economic activities that have similar or larger impacts on deforestation and land use (see discussion in Wicke *et al.,* 2011).

In this study, we document land cover and land use change in three palm oil producing countries (Figure 1): Indonesia, Malaysia, and Papua New Guinea between 1990 and 2010. We seek to identify patterns and trends in the development of oil palm plantations in these countries and to document the effects, extent, distribution, and rate of growth of this globally important commodity on forest landscapes and peat soils, as well as to document the conversion and use of other types of land cover as a source of new oil palm plantations.

Figure 1. Map of the study area, including Indonesia, Malaysia and Papua New Guinea.

MATERIALS AND METHODS

This study focuses on the three principal palm oil producing countries in Southeast Asia and the Pacific Region: Indonesia, Malaysia and Papua New Guinea (Figure 1). In Indonesia, only three main regions were evaluated: Sumatra, Kalimantan (the Indonesian part of the island of Borneo) and Papua (formerly Irian Jaya), which were targeted because the overwhelming majority of palm oil in Indonesia is produced in these provinces. In Malaysia, analysis included the entire country, but was stratified by region: Peninsular Malaysia, Sabah and Sarawak. The study of Papua New Guinea covered the entire country, including the island of New Britain.

This study distinguishes between mineral and peat soils, and within each of these two broad categories, we stratified land cover and land use change for both natural and human altered land cover types, including both productive and so-called degraded land. In Indonesia and Malaysia, landscapes were evaluated over three temporal periods (1990-2000, 2001-2005, and 2006-2009/2010), but in Papua New Guinea land use change was documented for two periods only (1990-2000 and 2001-2009/2010). Mixed data sets were used for 2009 and 2010 due to the scarcity of cloud-free Landsat images for 2010.

The study focused largely on large-scale oil palm plantation complexes that include both corporate and associated (schemed) smallholder plantings; we probably exclude most independent smallholders, whose oil palm plantings are mixed with other crops or trees and which lack obvious spatial patterns necessary for their identification using satellite imagery. In Indonesia, smallholdings are reported to comprise about 40% of the total area dedicated to oil palm (Jelsma *et al.,* 2009), but that value has not been validated by remote sensing studies, nor is it clear what percentage of this area is composed of schemed and independent smallholders. In this study, all oil palm plantations visually identified on the images were aggregated within the polygons.

Landsat 4, 5, and 7 satellite images were viewed using ArcGIS® software and subject to on-screen analysis and differentiation for the land cover types and land uses (Table 1). On screen analysis to directly indentify land cover types relies on the computer mouse as a tracing instrument, which differs from image analysis that classifies individual pixels using mathematical algorithms based on reflectance values of individual pixels (e.g., Hansen *et al.,* 2009; Carlson *et al.,* 2012b). Images covering Indonesia were geometrically corrected using the Forestry Thematic Base system (*Peta Dasar Thematik Kehutanan)* of the Ministry of Forestry of Indonesia (MOFRI, 2008). For Malaysia and Papua New Guinea, images were geo-referenced to previously orthorectfied Landsat images that were downloaded from NASA data distribution web sites.

We distinguished 22 different land cover types adapted from criteria used by MOFRI and the Ministry of Agriculture of Indonesia (MOARI). Land cover classes were delineated based on the classification systems used by MOFRI for Sumatra and Papua and by MOARI for Kalimantan. The MOFRI and MOARI systems use 22 and 21 classes respectively. The two classifications are similar, but MOARI recognizes rubber plantations, which is included within the crop plantation class of the MOFRI system, while grassland and swamp grassland classes in MOARI are classified as shrub in the MOFRI classification. We harmonized the classification systems to create a slightly modified version composed of 22 classes that reflect differences in above ground carbon stocks and which recognizes a specific oil palm plantation category (Table 1). The land cover classification used by Malaysian authorities is based on a different set of criteria (Rashid *et al.,* 2013 – this publication); consequently, for the purpose of comparability and with the goal of producing a single sector-wide study, we conducted an analysis for Malaysia and Papua New Guinea using the same methodological approach and classification system applied to Indonesia with 22 different land cover types.

We used a multistage visual technique based on an on-screen interpretation to directly digitize land cover units (Figure 2). We displayed the images as false color composites using Landsat bands: 3 (0.63-0.69 µm, red), 4 (0.76-0.90 µm, near infrared) and 5 (1.55-1.75 µm, mid-infrared); the combination of the selected channels was displayed on the screen according to the scheme with bands 5-4-3 displayed as red, green and blue, respectively. To assist in the interpretation and to validate the final product, technicians compared images with high resolution images from Google Earth, when available. In addition, images were overlaid with other layers of information, such as population centers, roads and existing administrative boundaries and a previously conducted study of land cover change in Papua and Riau (Tropenbos, unpublished).

The spatial distribution and extent of peat soils was obtained from Wetlands International for Indonesia (Wahyunto & Suryadiputra, 2008) and from a Harmonized World Soil Map for Malaysia (FAO, 2009), which were used to guide the delineation of swamp forest and other wetland habitats. Nonetheless, the identification and delineation of swamp forest, swamp shrub and swamp grassland were based on multiple criteria, which included the spectral and spatial attributes of the satellite images, as well as the landscape context of the area being evaluated (Table 1). Although there is considerable overlap, swamp categories were not entirely nested within the peat polygon. Consequently, data summaries for the four swamp vegetation classes (undisturbed swamp forest, disturbed swamp forest, swamp shrub land and swamp grassland) include both mineral and peat soils; however, data summaries for peat soils were constrained by the Wetlands International peat soil map polygons.

Figure 2. Flow chart showing the steps in the land cover change analysis.

The primary objective of the study was to document land use for the palm oil sector; however, we also analyzed land cover changes among all 22 land cover classes. The primary output of the data analysis was a 22 x 22 land cover change matrix, which was subsequently used to drive the models that estimate the GHG emissions linked to land use change (see Agus *et al.,* 2013 – this publication). However, to better understand the dynamics of oil palm plantation development and facilitate communication of the results, the output of the land use change analysis was also organized according to aggregate land cover classes based on i) the degree of intervention/disturbance (natural versus productive), ii) hydrology (upland versus wetland), iii) degree of disturbance and iv) type of agriculture or forestry production (Table 2). The results were compared with land cover maps and published statistics from other studies (Hansen *et al.,* 2009; Broich *et al.,* 2011; Miettinen *et al.,* 2012a; SarVision 2011; Ekadinata & Dewi, 201; Rashid *et al.,* 2013 – this publication) as well as with statistics published by the palm oil sector and government agencies.

Table 1. Synchronized land cover classification ranked based on above ground carbon stocks.

Code	Land Cover Type	Description and Landscape Context	Attributes when spectral bands are displayed in false color composite: red (5), green (4), blue (3)
UDF	Undisturbed Upland Forest	Natural forest, highly diverse species and high basal area. Well drained soils, often on hilly or mountainous terrain; absence of logging roads or settlements.	Reflectance: Medium in band 4, Medium to low in bands 5 and 3 (strong green). Texture: irregular and conspicuous due to canopy heterogeneity (pixels ranging from light to dark green).
DIF	Disturbed Upland Forest	Same as above, but basal area reduced significantly due to logging. Evidence of logging, including roads and small clearings typical of logging platforms.	Reflectance: Similar to UDF, with greater reflectance in all bands (strong green), but brighter in comparison to UDF. Texture: strongly contrasting due to greater reflection in all bands from isolated pixels impacted by logging (yellow to green - speckled appearance).
SCH	Upland Shrub land	Open woody vegetation, often as part of a mosaic including forest and grassland. Well drained soils on a variety of landscapes impacted by fire and logging; previous temporal periods reveal forest (UDF) or disturbed forest (DIF).	Reflectance: High to medium in band 4, 5 and 3 (whitish, to light green to yellow). Texture; Rough and irregular with periodic dark patches (disturbed forest remnants) and light patches (grassland).
GRS	Upland Grassland	Open vegetation dominated by grasses (most often *Imperata*). Upland, well drained soils often in association with shrub land.	Reflectance: Very high in bands 4, 5 and 3 (light green to tan or grey). Texture: Smooth and uniform.
USF	Undisturbed Swamp Forest	Natural forest with temporary or permanent inundation. Associated with peat domes and meandering rivers in coastal areas; absence of logging canals.	Reflectance: Medium in band 4, 5 and medium low in band 3 (dark green). Texture: smooth to irregular dark (green to dark green).
DSF	Disturbed Swamp Forest	Same as USF. Evidence of logging, regular network of canals and small-scale clearings.	Reflectance: Similar to USF, but with greater reflectance in all bands (light green color). Texture: smooth due to homogeneous canopy (light green to dark green).
SSH	Swamp Shrub land	Open woody vegetation on poorly drained soils; less than 3-6 m in height. On landscapes impacted by fire and logging in areas subject to temporary or permanent inundation; previous temporal periods reveal swamp forest (USF) or disturbed swamp forest (DSF).	Reflectance: High in band 4 and 5, and medium low in band 3 (white, to light green to yellow). Texture: rough and irregular, with periodic dark pixels (water or fire scars) and light patches (grassland).
SGR	Swamp Grassland	Extensive cover of herbaceous plants with scattered shrubs or trees. Inundated floodplains or impacted peat domes. Comparison with previous temporal periods revealed forest habitat.	Reflectance: Very high in bands 4, 5 and 3 (tan or grey to dark brownish pink). Texture: Smooth and uniform
TPL	Timber Plantation	Large industrial estates planted to timber or pulp species (e.g. *Gmelina* sp., *Paraserianthes falcataria, Acacia mangium*); canopy cover is around 30-50%. Regular geometry, typically in patches greater than 100 hectares; in association with road network and settlements located within forest area.	Reflectance: Medium to high in bands 4, 5 and 3, but with greater variance in reflectance in all bands (purple green color). Texture: smooth due to homogeneous canopy (light green to dark green).

Table 1. Synchronized land cover classification (continued).

Code	Land Cover Type	Description and Landscape Context	Attributes when spectral bands are displayed in false color composite: red (5), green (4), blue (3)
MTC	**Mixed Tree Crops / Agroforest**	Mosaic of cultivated and fallow land, usually located within 0.5-1 km of settlement or road; canopy cover between 5 and 60%; assumed to be small-scale plantings of a range of commercial species. Irregular geometry associated with primary and secondary road networks; comparison with past temporal periods revealed similar pattern on same or nearby landscapes.	Reflectance: Medium to high in bands 4, 5 and 3 (light green to yellow green). Texture: Smooth with periodic dark patches (forest remnants) and light patches (crops/settlements).
RPL	**Rubber Plantation**	Well drained landscapes of variable topography with large to medium sized industrial estates planted to rubber (*Hevea brasiliensis*). Regular geometry, typically in patches greater than 100 hectares; in association with road network.	Reflectance: Moderate to high reflectance in band 4, 5 and 6 (light green to green). Texture: smooth due to very homogeneous canopy of monoculture.
OPL	**Oil Palm Plantation**	Large industrial estates planted to oil palm; canopy cover variable depending on age. Regular geometry characterized by discernible rows and internal plantation road network, typically in patches greater than 1000 hectares.	Reflectance: Medium to high band 4, 5 and 3 (light green to green). Texture: smooth due to homogenous canopy indicating monoculture.
BRL	**Bare Soil**	Bare rock, gravel, sand, silt, clay, or other exposed soil. Often includes recently cleared (deforested) areas, landscapes impacted by fire and portions of estates undergoing replanting procedures.	Reflectance: High in band 4, medium to low in bands 3 and 5 (tan to brown to red). Texture: smooth.
DCL	**Dry Cultivated Land**	Open area characterized by herbaceous vegetation intensively managed for row crops or pasture. Associated with road networks and human settlements.	Reflectance: High in bands 4, 5 and 3 (bright to dark tans and browns, with blue and pink spots). Texture: smooth and uniform, but with dark patches depending on crop cycle.
RCF	**Rice Field**	Open area characterized by herbaceous vegetation (rice paddy), with seasonal or permanent inundation. Reticular patterns of dikes and canals, usually in association with settlements.	Reflectance: Low in band 4, very low in bands 5 and 3 (high absorbance from water – depending on season; (blue to blackish color). Texture: smooth and uniform pattern.
SET	**Settlements**	Villages, urban areas, harbors, airports, industrial areas, open mining; typically associated with road network.	Reflectance: High to very high in bands 4, 5 and 3 (light red to straw colored). Texture: rough due to heterogeneity from buildings, exposed soil, and home gardens.
MIN	**Mining**	Open area with surface mining activities. Irregular, in association with settlements or industrial facilities.	Reflectance: High in bands 3, 4 and 5. (white to light blue). Texture: smooth.
UDM	**Undisturbed Mangrove**	Forest habitat near coast with high density of mangrove tree species in irregular patterns. Featuring temporary or permanent inundation in coastal and estuarine areas.	Reflectance: Low in bands 4, 5 and 3 (dark green). Texture: smooth due to homogenous canopy, but usually in association with water (purple green to dark green).

Table 1. Synchronized land cover classification (continued).

Code	Land Cover Type	Description and Landscape Context	Attributes when spectral bands are displayed in false color composite: red (5), green (4), blue (3)
DIM	Disturbed Mangrove	Same as UDM. Evidence of clearing and often in association with coastal fish ponds (CFP, see below).	Reflectance: Similar to UDM, but with greater variance in reflectance in all bands (purple green color). Texture: smooth due to homogeneous canopy (light green to dark green).
CFP	Coastal Fish Pond	Permanently flooded open areas. Reticular patterns in coastal areas; comparison with previous temporal periods often showed as DMF or UMF	Reflectance: Very low in all bands (black, dark blue or dark brown). Texture: smooth.
WAB	Water bodies	Rivers, streams and lakes. Identified in satellite images by high absorbance in all spectral bands; featuring temporary or permanent inundation, as evidenced in band 4.	Reflectance: Very low in all bands (dark blue to black). Texture: smooth.
NCL	Not Classified; Cloud	Not classified due to cloud cover.	Reflectance: Very high in all bands Texture: irregular to smooth.

Table 2. Aggregate land cover classes used to facilitate the communication of results.

Superior Class	Aggregate Class	Land Cover Types Codes
Upland habitats	Undisturbed upland forest	UDF
	Disturbed upland forest	DIF
	Upland shrub and grassland	SCH + GRS
Swamp habitats	Undisturbed swamp forest	USF
	Disturbed swamp forest	DSF
	Swamp shrub and grassland	SSH + SGR
Productive land use types	Agroforest, rubber and timber plantations	MTC + CPL +TPL
	Oil palm	OPL
	Intensive Agriculture	DCL +RCF
	Bare soil	BRL
Others	Others	SET + MIN + NCL +CFP + UDM+DIM+WAB

RESULTS

The multi-temporal analysis spanning from 1990 to 2010 documents the expansion of industrial scale oil palm plantations in Indonesia (Sumatra, Kalimantan and Papua), Malaysia and Papua New Guinea (Figures 3 and 4). Oil palm plantations in these regions grew from 3.5 Mha in 1990 to 13.1 Mha in 2010 (Table 3). The historical trend in oil palm plantation development in the region has stayed remarkably steady at around 7% annual growth rate over twenty years (Table 4).

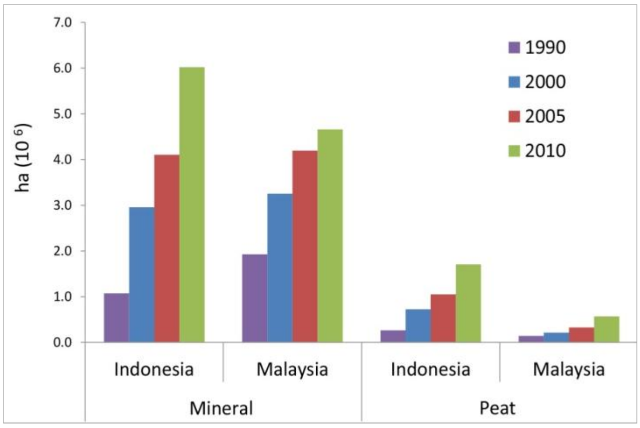

Figure 3. Oil palm plantation development on mineral soil and peat soil between 1990 and 2010 in Indonesia (Sumatra, Kalimantan and Papua), Malaysia and Papua New Guinea.

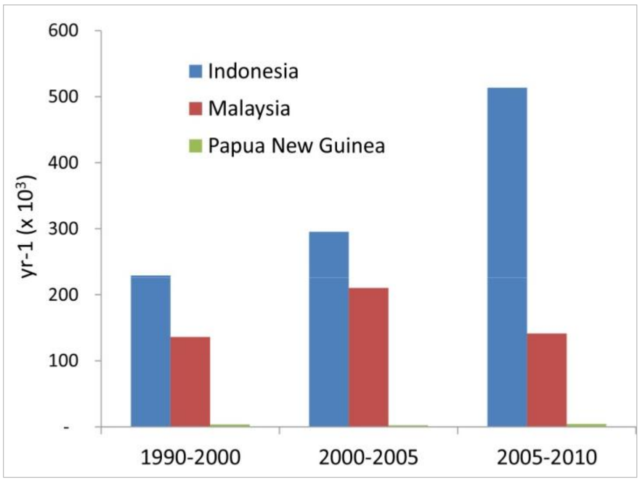

Figure 4. Annual growth in oil palm plantations in the major oil palm regions of Indonesia (Sumatra, Kalimantan and Papua), Malaysia and Papua New Guinea.

Table 3. Area (10^3 ha) of oil palm plantations in Indonesia, Malaysia, and Papua New Guinea 1990-2010.

Country	1990	2000	2005	2010
Indonesia	1,337	3,678	5,155	7,724
Malaysia	2,118	3,467	4,521	5,230
Papua New Guinea	57	91	103	134
Total	**3,511**	**7,236**	**9,780**	**13,087**

In the Indonesia study areas, the extent of oil palm plantations reached 7.7 Mha by 2010, of which 78% was found on mineral soils and 22% occurred on peat soils. In Malaysia, the total extent of oil palm plantations reached 5.4 Mha by 2010, where 87% occurred on mineral soil and 13% on peat soils (Figure 3). In Papua New Guinea, oil palm plantations reached 134,000 ha, all of which occurred on mineral soils.

Oil palm plantation development in Indonesia is mainly located in two regions: Sumatra and Kalimantan. In 2010, the total palm oil plantation area in Sumatra accounted for 4.7 Mha, while in Kalimantan the total oil palm plantation area accounted for 2.9 Mha. The expansion of oil palm plantations in Indonesia showed robust growth throughout all three temporal periods, while in Malaysia expansion was more moderate and showed a marked reduction in the rate of growth in the last temporal period (Figure 4).

Over the twenty year period between 1990 and 2010, approximately 36.5% of all oil palm plantations came from forest landscapes, including both upland and swamp habitats; nonetheless, only a small fraction of that conversion occurred on undisturbed forest landscapes (Table 5). Only 0.1% of oil palm plantations were sourced from undisturbed upland forest, while undisturbed swamp forest contributed 4.0% between 1990 and 2010. Development in Indonesia tended to follow a period of forest degradation, as evidenced by the large area of shrub land, grassland and agroforest habitats that were converted to oil palm plantations (Table 5). In contrast, conversion tended to be more direct in Malaysia with the conversion of disturbed forest for oil palm plantations, without large areas passing through a transitional stage of shrub land or agroforest that had been created in previous temporal periods via the degradation of forest landscapes.

Table 4. Mean annual growth rates of oil palm plantations for each country and sub-national unit; the values for the first temporal period are based on the mean of ten points generated by simple linear regression models, while the last two epochs are the mathematical average of the total change for the five year period.

	1990 - 2000 mean annual growth rate		2001 - 2005 mean annual growth rate		2006 - 2010 mean annual growth rate	
	10^3 ha yr^{-1}	%	10^3 ha yr^{-1}	%	10^3 ha yr^{-1}	%
Indonesia	229	10.5	295	8.0	514	10.0
Sumatra	*167*	*9.0*	*219*	*9.0*	*151*	*3.8*
Kalimantan	*65*	*21.5*	*72*	*9.7*	*360*	*32.9*
Papua	*1.9*	*5.1*	*4.3*	*9.0*	*2.9*	*4.3*
Malaysia	135	6.4	211	6.1	142	3.1
Peninsular	*44*	*2.6*	*72*	*3.4*	*36*	*1.5*
Sabah	*64*	*9.7*	*73*	*7.4*	*30*	*2.2*
Sarawak	*27*	*16.5*	*66*	*19.9*	*75*	*11.4*
Papua New Guinea	3.4	6.1	2.4	2.7	4.3	4.7
Total	**373**	**7.0**	**509**	**7.0**	**662**	**6.8**

Table 5. Prior land use of all new plantations established between 1990 and 2010.

Aggregate Class	Indonesia 1990 - 2010		Malaysia 1990 - 2010		Papua New Guinea 1990 - 2010		Total 1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest	13	0.2			4.6	6.0	18	0.2
Disturbed Upland Forest	1,207	18.9	1,239	38.1	37	46.0	2,483	25.6
Upland Shrub & Grasslands	1,268	19.9	15	0.5	28	34.8	1,311	13.5
Undisturbed Swamp Forest	384	6.0	0.5	0.0			384	4.0
Disturbed Swamp Forest	539	8.4	126	3.9	0.2	0.2	665	6.9
Swamp Shrub & Grasslands	411	6.4	4.9	0.2	6.6	8.3	423	4.4
Agroforest & Plantation	2,176	34.1	1,119	34.4			3,295	34.2
Intensive Agriculture	212	3.3	8.5	0.3	3.9	5.2	224	2.3
Bare Soil	74	1.2	731	22.5			806	8.3
Others	102	1.6	7.3	0.3			108	1.1
Total new plantations	**6,387**		**3,252**		**80**		**9,718**	

Indonesia

Land covered with oil palm plantations in the three largest oil palm growing regions in Indonesia (Sumatra, Kalimantan, Papua) reached a total area of 7.7 Mha in 2010 (Figure 5). A relatively small area of oil palm plantations is found on the islands of Java and Sulawesi, but these were not documented by this study. Oil palm plantations have expanded consistently over the last 20 year period, fluctuating from 10.5% of annual growth in the first temporal period, declining slightly to 8% between 2001 and 2010, and then returning to 10% annual growth in the last five year period. In Kalimantan, growth in new plantations reached 33% annually between 2006 and 2010, but this growth was accompanied by a decline in the rate of expansion in Sumatra from 9% to 3.8%. Growth of the sector in Papua was relatively slow throughout the entire study period.

Although land cover change linked to oil palm was documented from 1990 to 2010, changes that involved other land cover types were evaluated only between 2001 and 2010 (Table 6 and 7). Between 2001 and 2005, the conversion of agroforest and plantations was important, which coincided with an era of expansion in Sumatra, a region that is characterized by greater levels of human disturbance and past land use change. Between 2006 and 2010, however, the growing importance of Kalimantan as an expansion zone led to an increase in the conversion of natural habitat types, including mostly disturbed forests, but also large areas of shrub and grassland habitats (Table 7). Low to moderate biomass land cover types, including shrub land in Kalimantan and agroforest in Sumatra, represent important transitional categories between disturbed, albeit intact, forests and productive land use types dedicated to agriculture or plantation estates. Overall, the total area for these transitional land use types do not change greatly between temporal periods, because the increase in area due to forest loss was offset by the conversion of these land cover types to oil palm or other forms of agriculture (see below and in Supplementary Material).

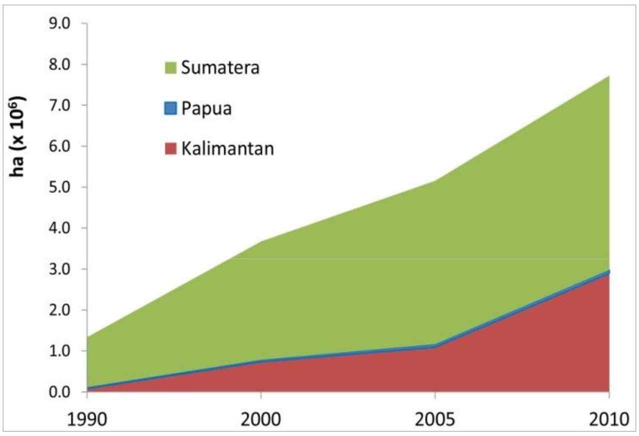

Figure 5. The expansion of oil palm plantations between 1990 and 2010 in the three major oil palm regions of Indonesia (Sumatra, Kalimantan and Papua).

Table 6. Land cover (10^3 hectares) in 2000, 2005, and 2010 in Sumatra, Kalimantan and Papua regions of Indonesia.

Aggregate classes	2000	2005	2010
Undisturbed Upland Forest	42,792	40,485	38,063
Disturbed Upland Forest	23,233	24,336	24,500
Upland Shrub and Grassland	17,399	16,141	15,927
Undisturbed Swamp Forest	10,160	9,791	9,014
Disturbed Swamp Forest	6,267	5,627	5,140
Swamp Shrub and Grassland	7,413	7,372	7,595
Agroforest and Plantation	15,053	14,421	13,762
Oil Palm Plantations	3,678	5,155	7,724
Intensive Agriculture	7,736	9,460	11,316
Bare soil	2,192	2,041	1,996
Others	6,862	7,955	7,749
Total	**142,785**	**142,785**	**142,785**

Table 7. Prior land use of all new oil palm plantations established in the three main oil palm regions of Indonesia (Sumatra, Kalimantan and Papua) between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 -2005		2005 -2010		1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest	1.3	0.1	2.6	0.2	8.7	0.3	13	0.2
Disturbed Upland Forest	475	20.3	88	6.0	644	25.1	1,207	18.9
Upland Shrub & Grasslands	370	15.8	217	14.7	681	26.5	1,268	19.9
Undisturbed Swamp Forest	374	16.0	0.4	0.0	9.5	0.4	384	6.0
Disturbed Swamp Forest	174	7.4	78	5.3	288	11.2	539	8.4
Swamp Shrub & Grasslands	60	2.6	32	2.2	319	12.4	411	6.4
Agroforest & Plantation	824	35.2	977	66.2	375	14.6	2,176	34.1
Intensive Agriculture	17	0.7	56	3.8	139	5.4	212	3.3
Bare Soil	6.1	0.3	21	1.4	48	1.9	74	1.2
Others	40	1.7	5	0.1	57	2.2	102	1.5
Total New Plantations	2,341		1,477		2,569		6,387	

Sumatra

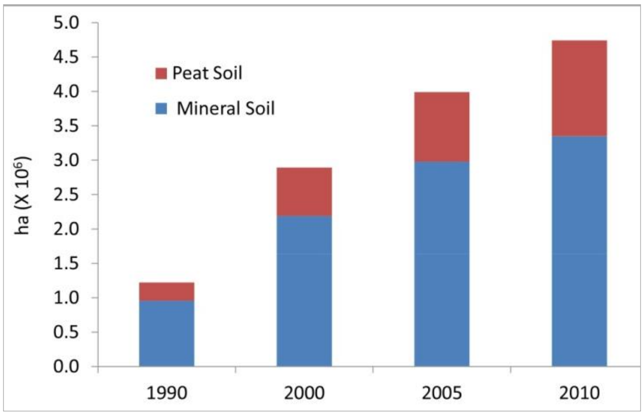

Figure 7. The area planted to oil palm in 1990, 2000, 2005 and 2010 on mineral and peat soil in Sumatra.

Sumatra has the oldest and most mature oil palm plantations in Indonesia, which reached a total of 4.7 Mha by 2010, representing 10% of its total land area (Table 8). About 3.4 Mha have been established on mineral soils occupying about 8% of the existing mineral soil land bank, while 1.4 Mha occur on peat soils representing 19.4% of the total peat soil area (Figures 6 and 7). Between 2001 and 2005, oil palm plantation development overwhelmingly occurred due to the conversion of agroforest and rubber plantations, but in the more recent period, the sources of land for development were more diverse. Although the conversion of agroforest and rubber plantations remained the largest source of land for development, more than 866,000 ha of disturbed and undisturbed swamp forest, as well as open swamp habitat (i.e., highly degraded swamp forest) were converted to oil palm plantations (Table 9). In Sumatra, the category bare soil was used largely by GIS technicians for grouping permanently bare soils and did not impact the land use change statistics related to oil palm plantations (see Supplementary Material).

Table 8. Land cover area (10^3 ha) in 2000, 2005, and 2010 in Sumatra.

Aggregate Class	2000	2005	2010
Undisturbed Upland Forest	5,753	5,749	5,321
Disturbed Upland Forest	5,904	5,757	5,686
Upland Shrub and Grassland	4,821	3,403	3,623
Undisturbed Swamp Forest	550	543	467
Disturbed Swamp Forest	3,109	2,519	2,073
Swamp Shrub and Grassland	3,014	2,798	2,681
Agroforest and Plantation	13,432	12,679	12,012
Oil Palm Plantations	2,893	3,990	4,743
Intensive Agriculture	4,554	5,658	6,700
Bare soil	1,364	1,177	1,194
Others	2,396	3,518	3,291
Total	47,791	47,791	47,791

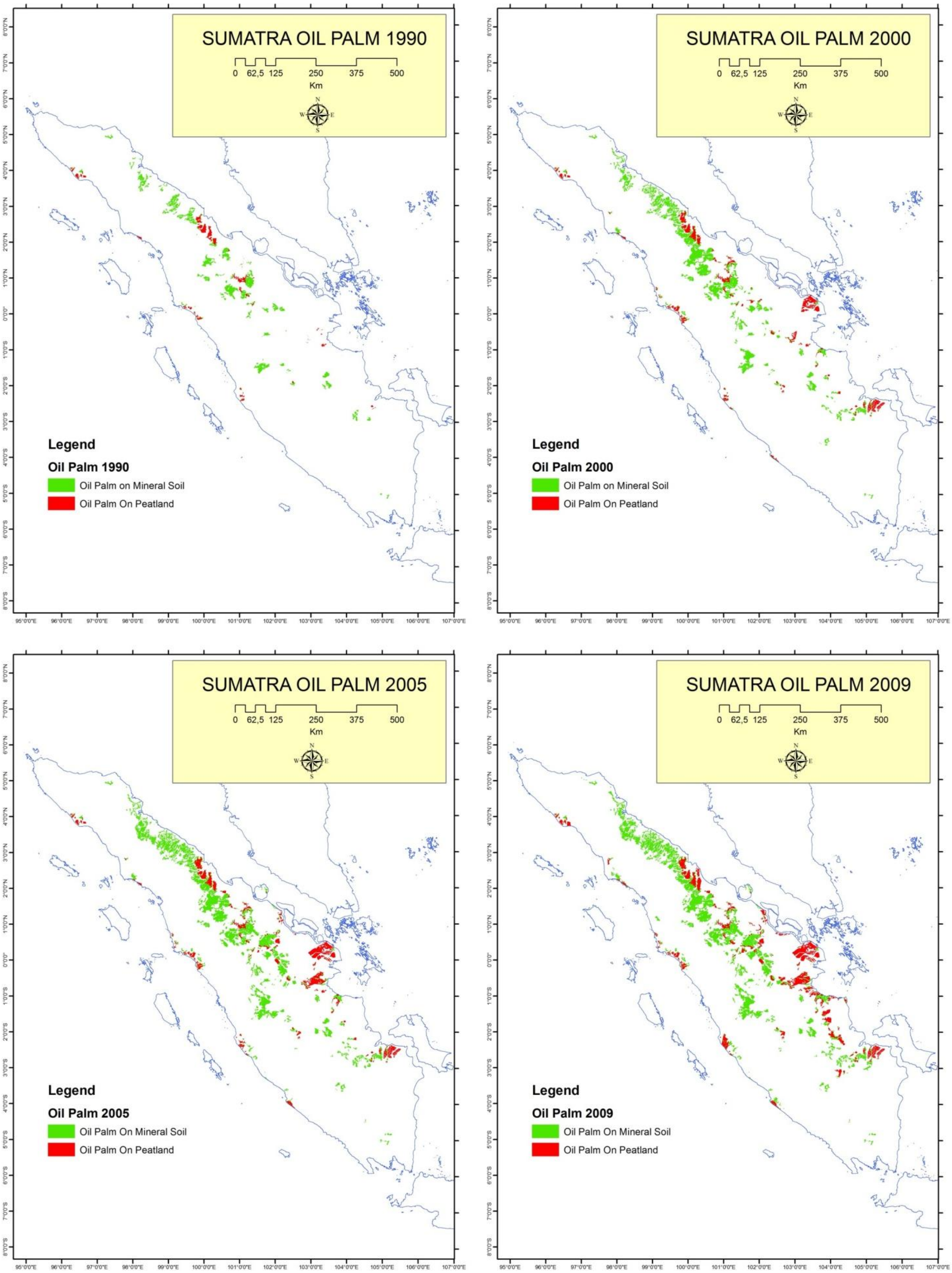

Figure 6. The expansion of oil palm plantations in Sumatra between 1990 and 2009.

Table 9. Prior land use of all new oil palm plantations established in Sumatra between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 -2005		2005 -2010		1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest								
Disturbed Upland Forest	170	10.2	11	1.0	28	3.7	209	5.9
Upland Shrub & Grasslands	115	6.9	24	2.2	6	0.8	145	4.1
Undisturbed Swamp Forest	364	21.8	0.3	0.0	7	0.9	370	10.5
Disturbed Swamp Forest	142	8.5	53	4.8	108	14.4	303	8.6
Swamp Shrub & Grasslands	56	3.4	11	1.0	126	16.7	192	5.5
Agroforest & Plantation	813	48.7	936	85.3	321	42.6	2,070	58.8
Intensive Agriculture	6.1	0	37	3.3	54	7.1	95	2.7
Bare Soil	4.8	0.3	21	1.9	47	6.2	74	2.1
Others			4.6	0.4	57	8.0	62	1.8
Total New Plantations	**1,671**		**1,097**		**753**		**3,521**	

Kalimantan

Kalimantan has the second largest extent of oil palm plantations in Indonesia, most of which are concentrated in West Kalimantan, followed by Central Kalimantan, East Kalimantan, and South Kalimantan. The total areas dedicated to commercial oil palm plantations reached 2.9 Mha by 2010, or 5.4% of the total area of Kalimantan (Table 10); of these about 2.6 Mha were on mineral soils and 308,000 ha were on peat soils (Figure 8 and Figure 9). Undisturbed forests suffered progressive declines over both temporal periods, which corresponded to an increase in disturbed forest on both upland and swamp forest landscapes. The most notable categories of land cover, when compared to Sumatra and other regions, were shrub lands and grasslands that fluctuated slightly between 2000, 2005 and 2010 (Table 10). This was not a static land cover class, however, as large areas were converted to oil palm and other forms of agriculture, while an approximately equivalent area of forest was degraded by the non sustainable use of forest landscapes (Figure 10).

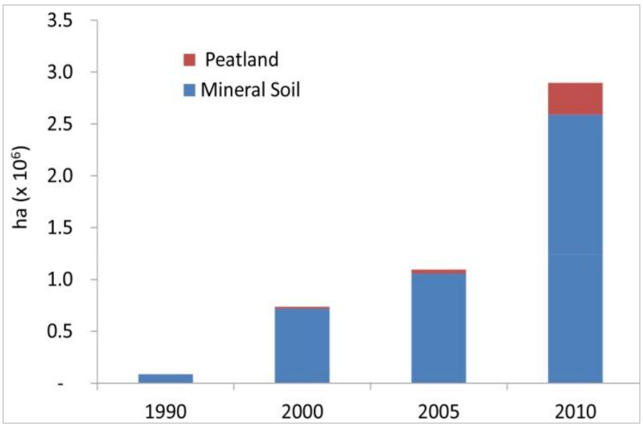

Figure 8. The area planted to oil palm in 1990, 2000, 2005 and 2010 on mineral and peat soil in Kalimantan.

Table 10. Land cover area (10^3 ha) in 2000, 2005, and 2010 in Kalimantan.

Aggregate Class	2000	2005	2010
Undisturbed Upland Forest	13,918	12,885	11,765
Disturbed Upland Forest	14,598	14,817	14,295
Upland Shrub and Grassland	10,967	11,043	10,551
Undisturbed Swamp Forest	2,677	2,690	2,306
Disturbed Swamp Forest	2,734	2,456	2,172
Swamp Shrub and Grassland	3,131	3,173	3,282
Agroforest and Plantation	1,359	1,497	1,492
Oil Palm Plantations	737	1,096	2,897
Intensive Agriculture	2,449	2,878	3,639
Bare soil	-	-	-
Others	1,171	1,208	1,342
Total	**53,742**	**53,742**	**53,742**

Approximately 48% of all oil palm plantations originated from the conversion of shrub or grassland habitats (40% upland and 8% swamp), which were followed closely by the direct conversion of forest habitat, representing about 44% of all new plantations (Table 11). The conversion of peat soils for oil palm increased over time, covering approximately 821 ha in 1990 (1% of all oil palm plantations) to more than 307,500 ha (11%) by 2010 (Figure 8). Similar to the trend observed in upland habitats in Kalimantan, the conversion of peat soils was also the consequence of a trajectory of land use characterized by the sequential degradation of undisturbed forest to disturbed forest to open swamp habitat prior to its conversion to oil palm (Figure 10). The category bare soil was not used by the GIS technicians for Kalimantan and has no impact on the summary calculations for land use.

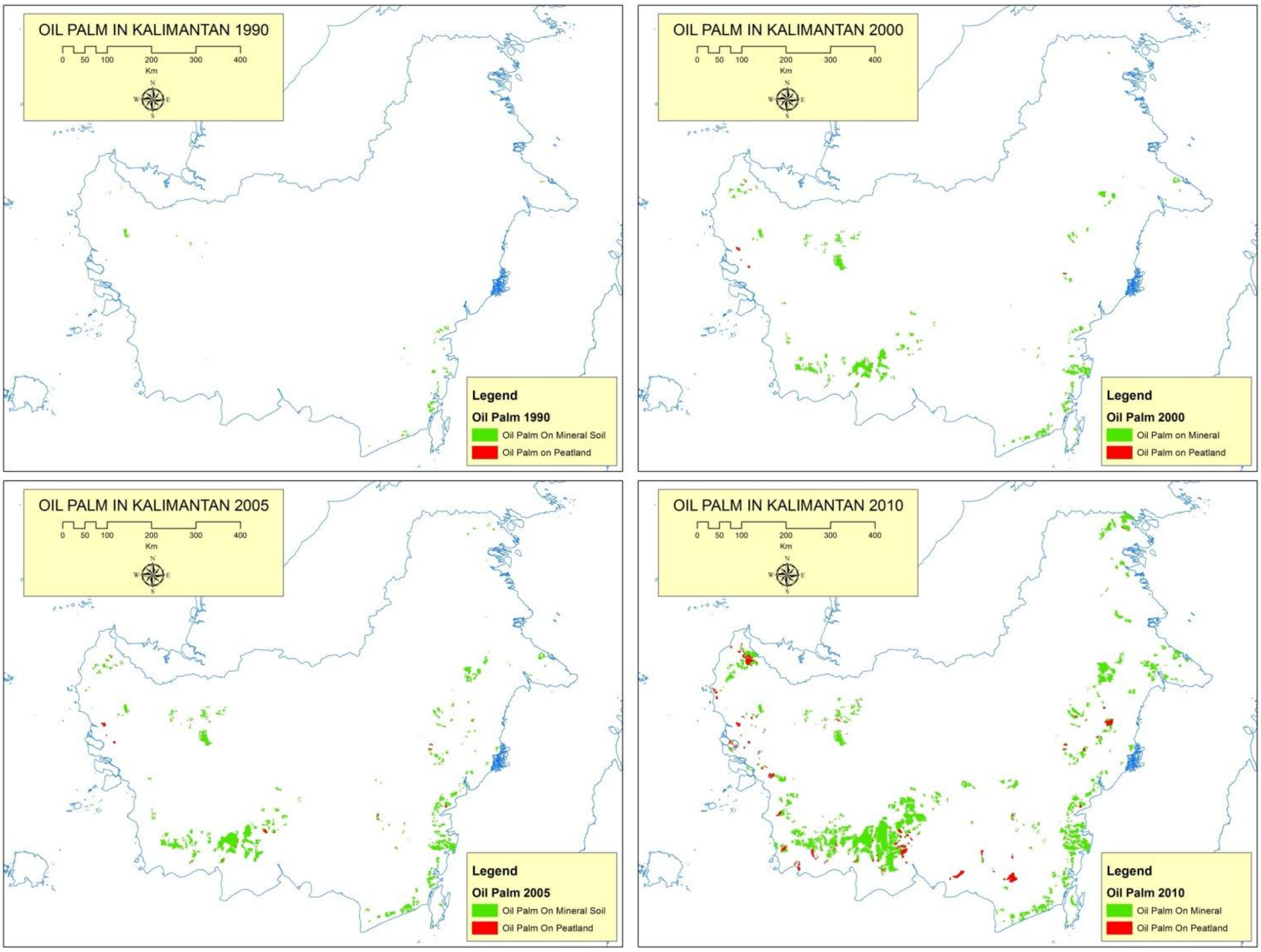

Figure 9. The expansion of oil palm plantations in Kalimantan between 1990 and 2010.

Table 11. Prior land use of all new oil palm plantations established in Kalimantan between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 -2005		2005 -2010		1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest	1.3	0.2	1.1	0.3			2.4	0.1
Disturbed Upland Forest	298	45.8	74	20.7	614	34	986	35.1
Upland Shrub & Grasslands	254	39.0	192	53.4	675	38	1,122	39.9
Undisturbed Swamp Forest					2.4	0.1	2.4	0.1
Disturbed Swamp Forest	32	4.9	25	6.9	179	10.0	236	8.4
Swamp Shrub & Grasslands	4.2	0.6	21	5.9	193	10.7	219	7.8
Agroforest & Plantation	9.2	1.4	27	7.4	52	2.9	88	3.1
Intensive Agriculture	12	1.9	19	5.4	85	4.7	116	4.1
Bare Soil								
Others	40	6.2		0.0			40	1.4
Total New Plantations	652		359		1,801		2,811	

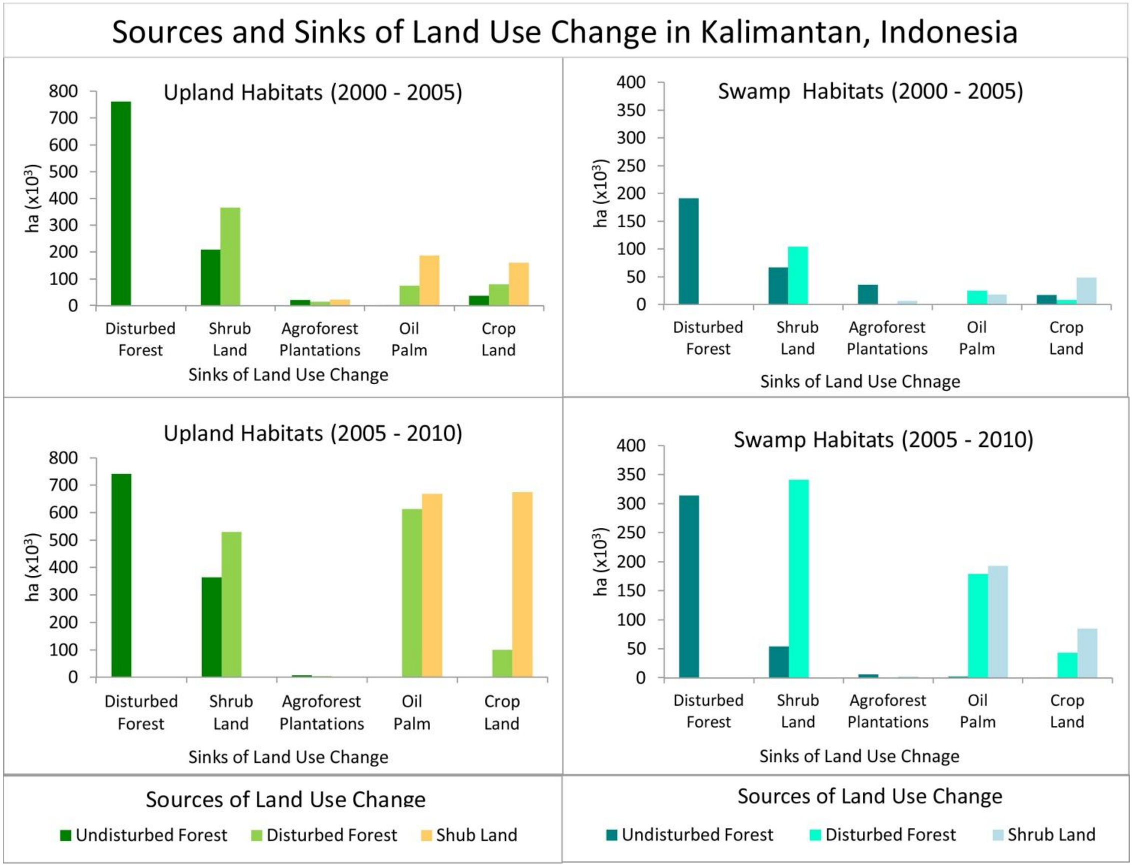

Figure 10. The conversion of forest in Kalimantan is a step-wise process where undisturbed forest is impacted by logging, which is sometimes followed by wildfire that further degrades areas into shrub land. The establishment of plantations or crops is largely the consequence of the conversion of disturbed forest or shrub land; this trajectory of degradation prior to conversion occurs on both upland and swamp habitats.

Papua (West Papua and Papua Provinces)

Total oil palm plantations in Papua reached 83,600 ha by 2010, with only about 1,700 ha located on peat soils (Figure 11 and 12). Unlike Sumatra and Kalimantan, the expansion of oil palm in Papua remains limited and more than 79% of the island remains covered by intact forest ecosystems (Table 12). Nevertheless, between 2000 and 2010, undisturbed upland forest in Papua declined by 2.1 Mha, while the extent of undisturbed swamp forest declined by about 690,000 ha (Table 12). Since the absolute numbers linked to the expansion of oil palm plantations are relatively small in any one temporal period, the relative contributions of the different land cover types to that expansion vary greatly (Table 13). When summed over the total twenty year period, the largest single source of new plantations was the aggregate category agroforest and other plantations; nonetheless, when the various forest categories are combined they sum to approximately 61% (Table 13).

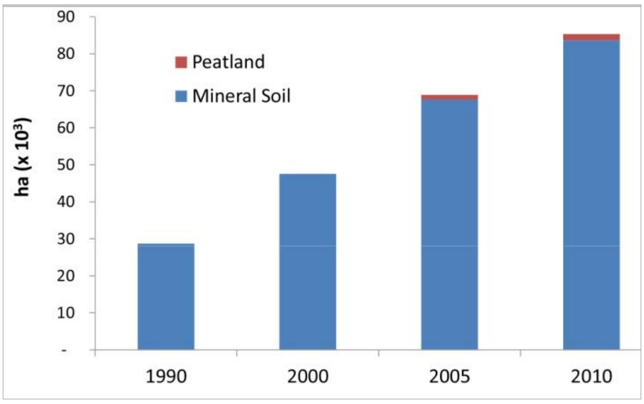

Figure 11. The area planted to oil palm in 1990, 2000, 2005 and 2010 on mineral and peat soil in Papua.

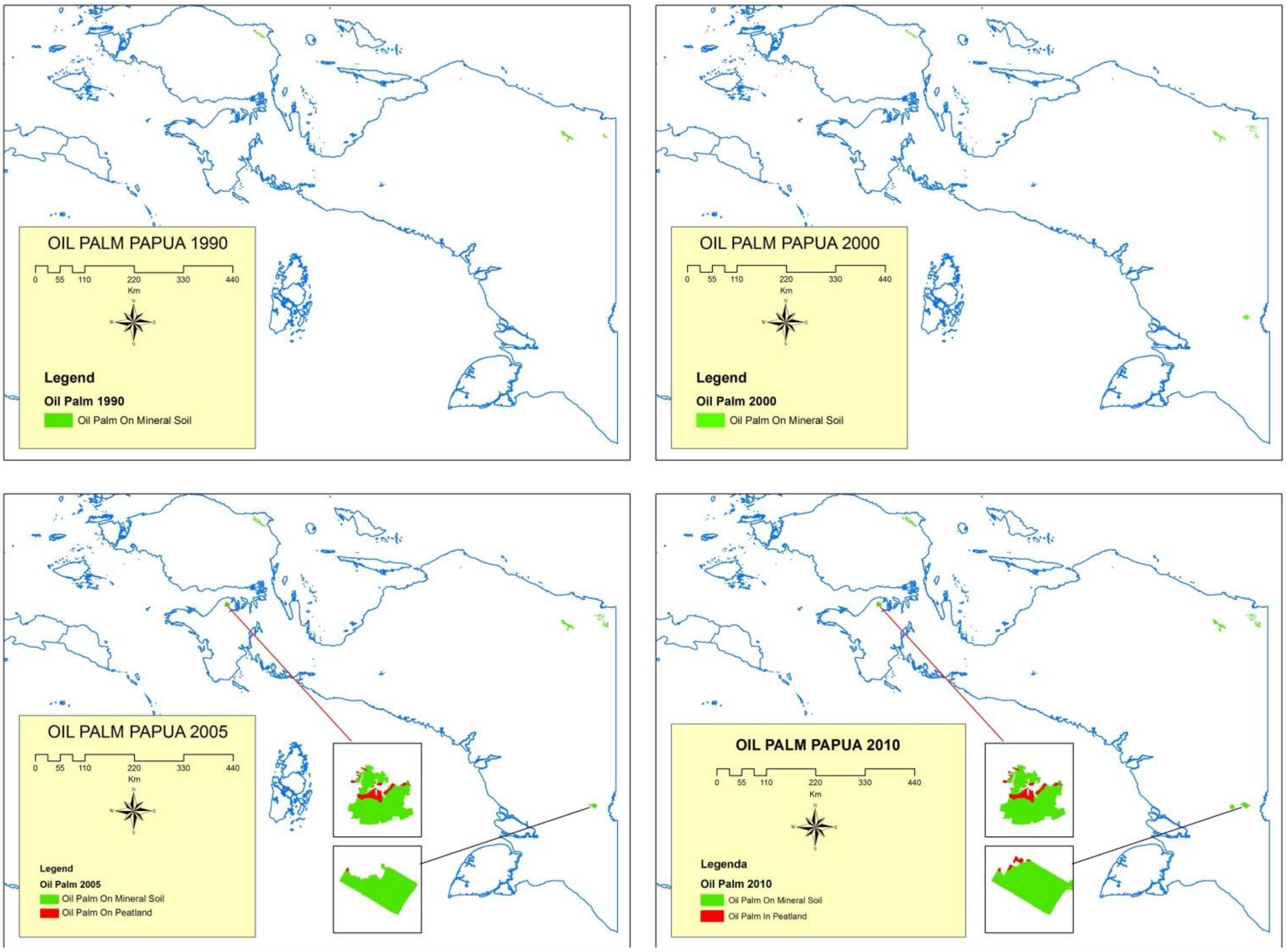

Figure 12. The expansion of oil palm plantations in Papua between 1990 and 2010.

Table 13. Prior land use of all new oil palm plantations in Papua between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 - 2005		2006 - 2010		1990 - 2010	
	ha	%	ha	%	ha	%	ha	%
Undisturbed Upland Forest	-		1,545	7.2	8,738	59.4	10,283	18.7
Disturbed Upland Forest	7,136	37.9	2,936	13.8	2,288	16	12,361	22.5
Upland Shrub & Grasslands	-		1,131	5.3	125	1	1,256	2.3
Undisturbed Swamp Forest	10,178	54.1	67	0.3	481	3	10,726	19.5
Disturbed Swamp Forest	-		53	0.2	137	0.9	191	0.3
Swamp Shrub & Grasslands	-		-		258	1.8	258	0.5
Agroforest & Plantation	1,505	8	14,876	69.7	2,064	14.0	18,446	33.6
Intensive Agriculture	-							
Bare Soil	-		156	0.7	621	4.2	778	1.4
Others	-		585	2.7	-	-	585	1.1
Total New Plantations	**18,820**		**21,350**		**14,713**		**54,883**	

Table 12. Land cover area (hectares) in 2000, 2005, and 2010 in Papua.

Aggregate Class	2000	2005	2010
Undisturbed Upland Forest	23,121	21,851	20,977
Disturbed Upland Forest	2,731	3,763	4,518
Upland Shrub and Grassland	1,611	1,695	1,753
Undisturbed Swamp Forest	6,932	6,557	6,241
Disturbed Swamp Forest	425	652	895
Swamp Shrub and Grassland	1,267	1,402	1,632
Agroforest and Plantation	262	245	257
Oil Palm Plantations	48	69	84
Intensive Agriculture	733	925	976
Bare soil	827	865	803
Others	3,295	3,230	3,116
Total	**41,252**	**41,252**	**41,252**

Malaysia

Malaysia has the second largest extent of oil palm plantations in the world, which in 2010 covered approximately 5.2 Mha or 16% of the total land area of the country, up from 6.4 % in 1990 (Table 14 and Figure 13). The rate of growth of new oil palm plantations has been decreasing over the past twenty years; it reached a high of 6.4% (134,926 ha) in the first period, but declined slightly to 6.1 % (210,261 ha) between 2001 and 2005 and then dropped to 3.1% (141,326 ha) annual growth in the last five year period (Table 15). Expansion during the first period was largely the result of the conversion of disturbed upland forest, followed by agroforest and plantations (Table 15), but during the second temporal period (2001-2005) the conversion of disturbed upland forest decreased markedly. In the last temporal period (2006-2010) the largest single source of young plantations was bare soil, which includes large areas of recently cleared forest in Sarawak and Sabah, as well as the conversion of rubber plantations and the renovation of older oil palm plantations in Peninsular Malaysia (see Supplementary Materials). To better understand the dynamics of the growth in oil palm development in Malaysia, we analyzed separately the expansion of oil palm plantations in the three geographical regions: Peninsular Malaysia, the state of Sabah, and the state of Sarawak.

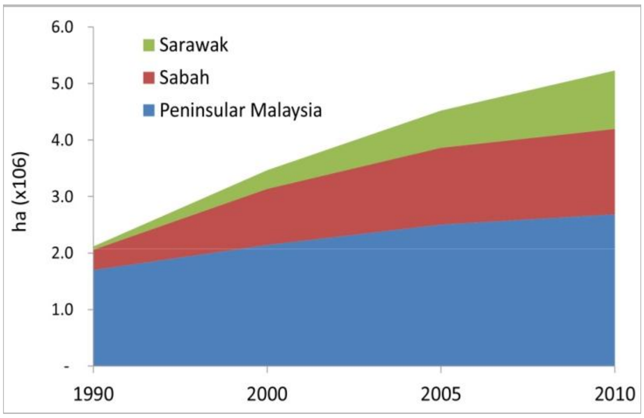

Figure 13. The expansion of oil palm plantations between 1990 and 2010 in Malaysia.

Table 14. Land cover area (10^3 ha) in Malaysia (Peninsular, Sarawak, Sabah) in 2000, 2005, and 2010.

Aggregate Class	2000	2005	2010
Undisturbed Upland Forest	3,710	3,153	3,150
Disturbed Upland Forest	16,901	16,875	16,340
Upland Shrub and Grassland	104	144	142
Undisturbed Swamp Forest	189	18	7
Disturbed Swamp Forest	928	904	746
Swamp Shrub and Grassland	44	46	51
Agroforest and Plantation	4,800	4,592	4,175
Oil Palm Plantations	3,467	4,521	5,230
Intensive Agriculture	684	681	682
Bare soil	624	527	937
Others	1,632	1,622	1,624
Total	**33,084**	**33,084**	**33,084**

Table 15. Prior land use of all new oil palm plantations established in all regions of Malaysia (Peninsular, Sarawak, Sabah) between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 - 2005		2006 - 2010		1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest			0.2	0.0			0.2	0.0
Disturbed Upland Forest	744	53.3	289	26	207	27.5	1,239	38.1
Upland Shrub & Grasslands	2.6	0.0	7.2	0.7	5.5	0.7	15	0.5
Undisturbed Swamp Forest	0.5	0.0					0.5	0.0
Disturbed Swamp Forest	36	2.6	46	4.2	43	5.8	126	3.9
Swamp Shrub & Grasslands	2.9	0.2	1.3	0.1	0.7	0.1	5.0	0.2
Agroforest & Plantation	511	37	380	34.3	228	30.4	1,119	34.4
Intensive Agriculture			8.4	0.8	0.1	0.0	8.5	0.3
Bare Soil	94	6.7	371	33.5	266	35.4	731	22.5
Others	3.9	0.3	3.2	0.3	0.1	0.0	7.0	0.2
Total New Plantations	**1,394**		**1,106**		**751**		**3,252**	

Peninsular Malaysia

The oldest oil palm plantations established in the country are located in Peninsular Malaysia with about 1.7 Mha existing in 1990, which by 2010 had increased to approximately 2.7 Mha, representing about 20% of the total area of the peninsula (Figure 14 and Table 16). There has been no direct conversion of undisturbed forest on the peninsula throughout the twenty year period, but the conversion of disturbed forest represented more than 38% of all new plantations in the first temporal period, but this then declined in the next two temporal periods (Table 17). The largest source of young oil palm plantations in all three temporal periods was from the conversion of agroforest and plantations. An evaluation of the bare soil category in the change matrix showed that a variable amalgam of different land types, including forest, rubber and oil palm plantations, were converted into this transitional category, prior to being replanted as oil palm plantations. The percentage of oil palm plantations on peat soils stayed relatively constant throughout, expanding proportionally with the sector, constituting about 8.1% of all oil palm plantations in 1990 and 7.9% in 2010 (Figure 15). If the forest conversion statistics are modified to reflect the proportion of bare soils that originated from forest habitats and that were allocated to oil palm plantations over the entire 20 year period, then approximately 28% of all plantations or 318,000 ha have originated due to forest conversion (see Supplementary Material).

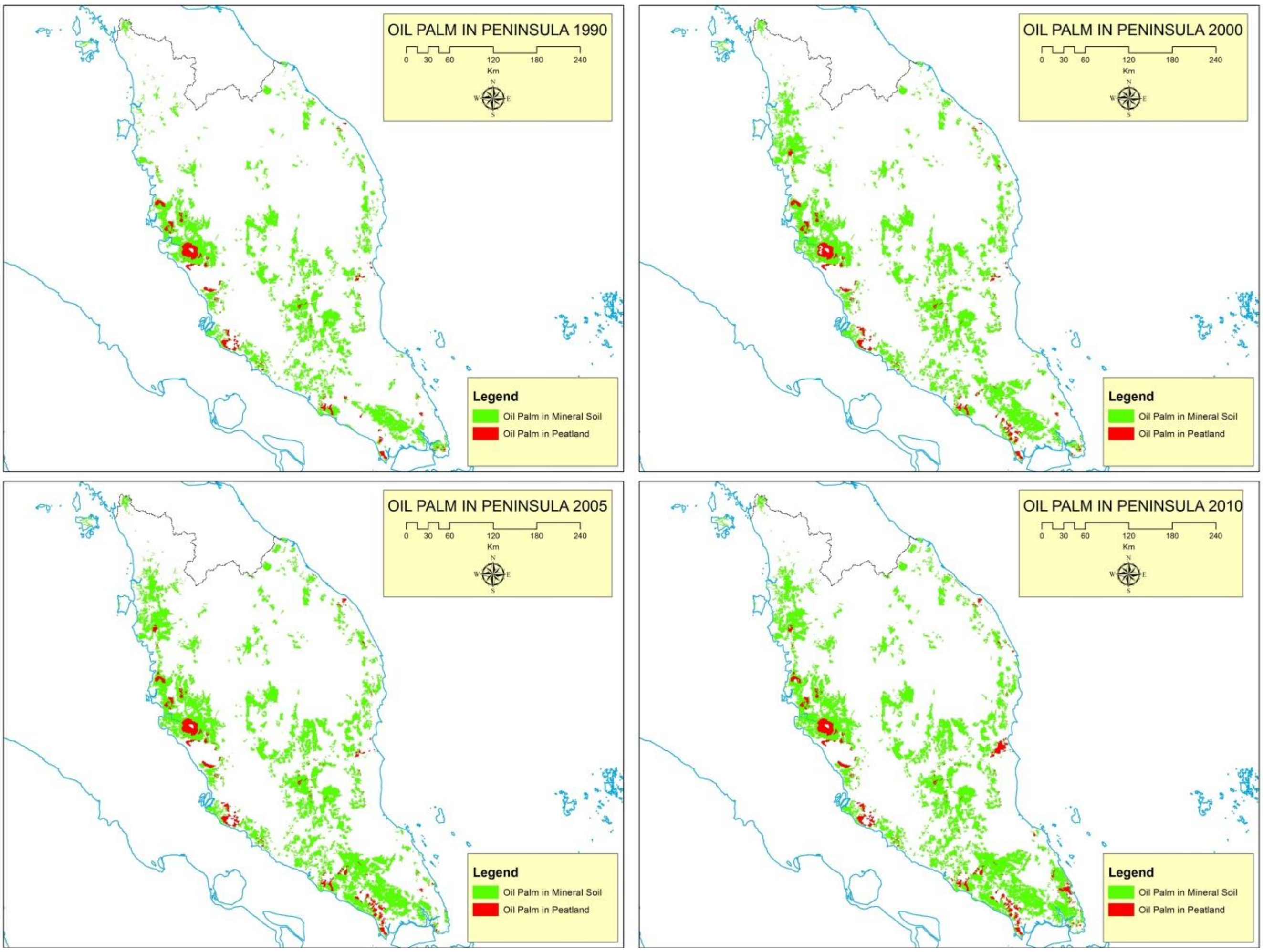

Figure 14. The expansion of oil palm plantations in Peninsular Malaysia between 1990 and 2010.

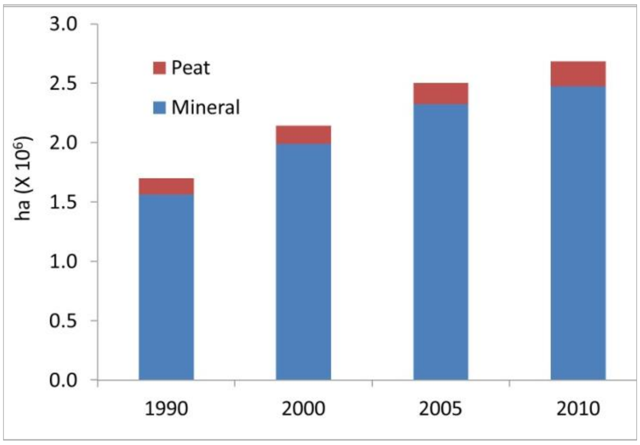

Figure 15. The area planted to oil palm in 1990, 2000, 2005 and 2010 on mineral and peat soil in Peninsular Malaysia.

Table 16. Land cover are (10^3 ha) in 2000, 2005 and 2010 in Peninsular Malaysia.

Aggregate Class	2000	2005	2010
Undisturbed Upland Forest	3,442	3,000	3000
Disturbed Upland Forest	2,416	2,670	2,578
Upland Shrub and Grassland	91	107	97
Undisturbed Swamp Forest	170	1	1
Disturbed Swamp Forest	270	346	338.99
Swamp Shrub and Grassland	22	22	23
Agroforest and Plantation	3,134	3,062	2,763
Oil Palm Plantations	2,144	2,504	2,686
Intensive Agriculture	359	355	357
Bare soil	306	287	511
Others	850	850	851
Total	**13,205**	**13,205**	**13,205**

Table 17. Prior land use of new oil palm plantations established in Peninsular Malaysia between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 - 2005		2006 - 2010		1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest								
Disturbed Upland Forest	174	35.5	63	15.4	14	6.2	251	22.4
Upland Shrub & Grasslands	0	0.0	6.6	1.6	4.9	2.2	12	1.0
Undisturbed Swamp Forest								
Disturbed Swamp Forest	9.7	2.0	20	4.9	0.1	0.1	30	2.7
Swamp Shrub & Grasslands								
Agroforest & Plantation	260	53.0	125	30.8	101	46.3	487	43.6
Intensive Agriculture	-	0.0	5.9	1.4			5.9	0.5
Bare Soil	43	8.8	185	45.4	99	45.2	327	29.3
Others	3.5	0.0	1.8	0.5			5.9	0.5
Total New Plantations	**491**		**407**		**219**		**1,118**	

Sabah

Sabah is situated on the northeast corner of Borneo with a total land area of 7.4 Mha; oil palm plantations covered 358,000 ha in 1990 and grew to more than 1.5 Mha by 2010 (Table 18). Most of this growth occurred between 1990 and 2000 when annual growth rates approached 10%; the rate of expansion then declined over time, and between 2006 and 2010 was 2.2% annually. By 2010, the area dedicated to oil palm corresponded to about 20% of the total area of Sabah (Figure 16). The largest single source of new plantations in Sabah over two decades has been disturbed, presumably logged, upland forest; nonetheless, during the period between 2001 and 2005, the conversion of agroforest and other types of plantations was nearly equivalent to the area converted from forest (Table 19). Sabah lacks extensive swamp forest formations and, consequently, the amount of oil palm on peat soils is minimal (Figure 17). If the forest conversion statistics are modified to reflect the proportion of bare soils that originated from forest habitats and that were allocated to oil palm plantations over the entire 20 year period, then approximately 62% of all plantations, or 714,000 ha, have originated due to forest conversion (see Supplementary Material).

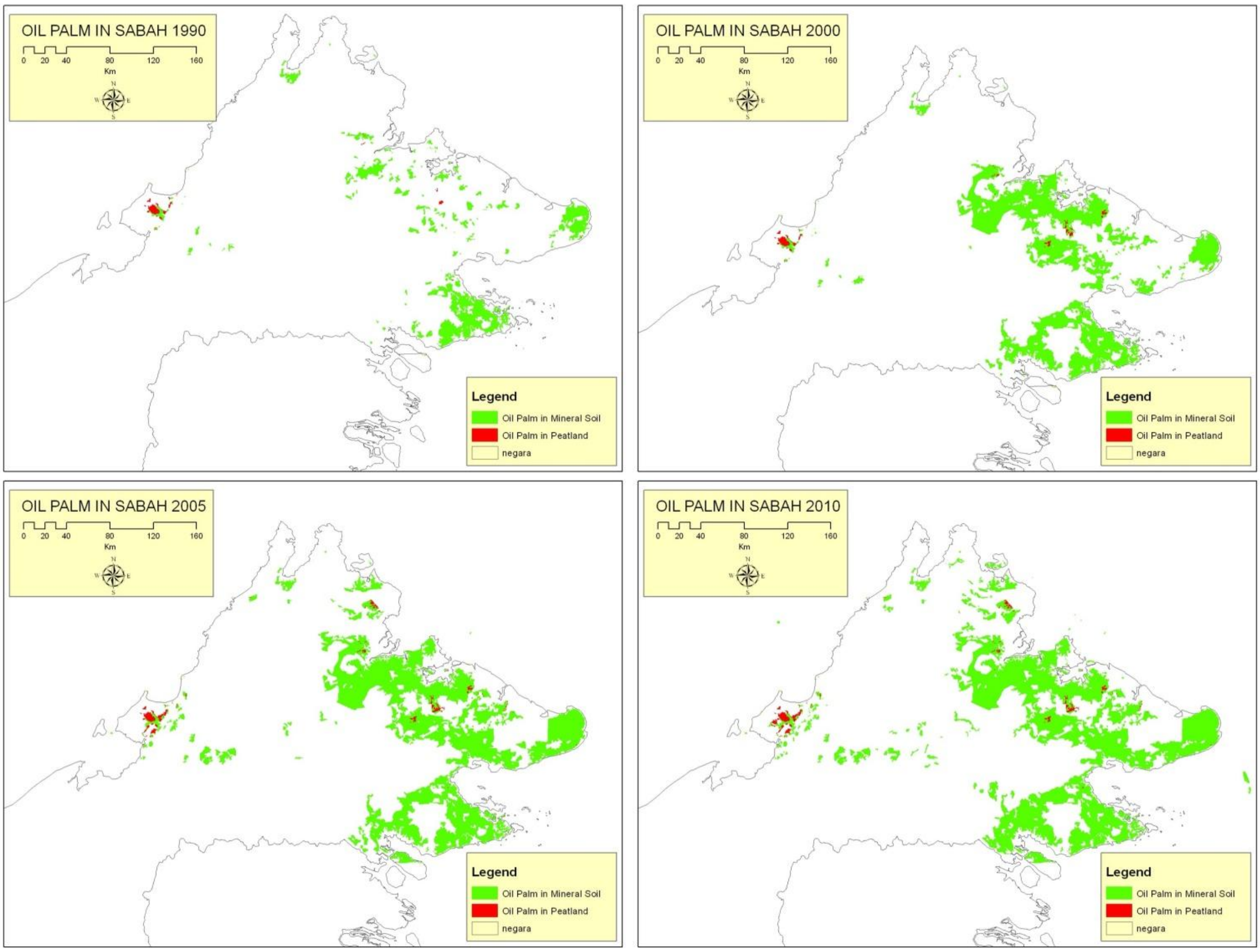

Figure 16. The expansion of oil palm plantations in Sabah between 1990 and 2010.

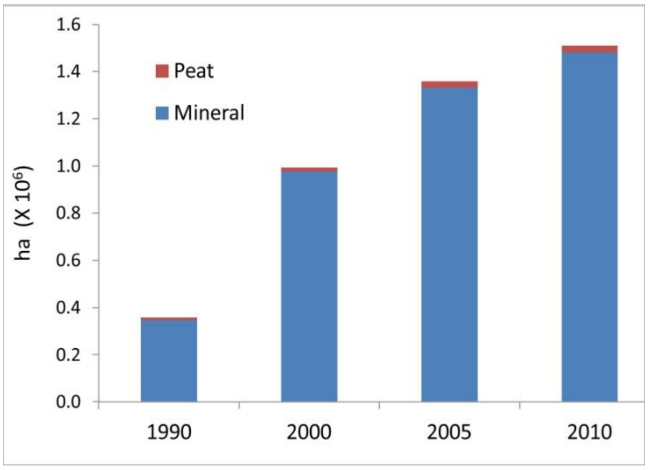

Figure 17. The area planted to oil palm in 1990, 2000, 2005 and 2010 on mineral and peat soil in Sabah.

Table 18. Land cover area (10^3 ha) in 2000, 2005 and 2010 in Sabah.

Aggregate Class	2000	2005	2010
Undisturbed Upland Forest	245	130	127.04
Disturbed Upland Forest	4,789	4,714	4,482
Upland Shrub and Grassland	7	9	10
Undisturbed Swamp Forest	17	15	4
Disturbed Swamp Forest	47	37	39
Swamp Shrub and Grassland	19	23	26
Agroforest and Plantation	404	350	405
Oil Palm Plantations	994	1,359	1,511
Intensive Agriculture	249	247	246
Bare soil	170	60	92
Others	489	487	487
Total	**7,431**	**7,431**	**7,431**

Table 19. Prior land use of all new plantations established in Sabah, Malaysia between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 - 2005		2006 - 2010		1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest	-	0.0	0	0.0	-	0.0	0	0.0
Disturbed Upland Forest	435	68.4	126	34.5	116	75.5	677	58.6
Upland Shrub & Grasslands	1.7	0.3	0.6	0.2	0.3	0.2	2.6	0.2
Undisturbed Swamp Forest	-	0.0	-	0.0	-	0.0	-	0.0
Disturbed Swamp Forest	5.0	0.8	6.2	1.7	0.6	0.4	12	1.0
Swamp Shrub & Grasslands	2.9	0.5	1.3	0.4	0.7	0.5	5.0	0.4
Agroforest & Plantation	163	25.7	125	34.2	7.8	5.1	296	25.6
Intensive Agriculture	-	0.0	0.4	0.1	-	0.0	0.4	0.0
Bare Soil	28	4.4	106	28.8	28	18.4	162	14.0
Others			0.4	0.1	-	0.0	0.4	0.0
Total New Plantations	**636**		**366**		**153**		**1,155**	

Sarawak

The development of the oil palm sector in Sarawak lagged behind both Sabah and Peninsular Malaysia; in 1990, the state had less than 61,000 ha of industrial scale plantations. Expansion occurred at the rate of 16.5% in the 1990s and 20% between 2001 and 2005 and remained at the relatively high level of 11.4% between 2006 and 2010. By 2010, the total extent of oil palm plantations had reached 1.03 Mha, or about 6% of the total area of Sarawak (Figure 18 and Table 20). The expansion of oil palm plantations has occurred largely as a consequence of the conversion of disturbed upland forest; other important sources of land cover include rubber and timber plantations and disturbed swamp forest (Table 21). Coastal Sarawak is characterized by large areas of peat swamp and the expansion of oil palm plantations on peat soils has become increasingly important over time; in 1990 only about 8% of the total oil palm plantation area was located on peat soils, but this value increased to more than 32% in 2010 (Figure 18 and 19).

As stated previously, the category identified as bare soil is a combination of the clearing of land cover types and examination of the change matrix for Sarawak reveals that approximately 29% originated from upland forest landscapes and 16% from swamp forest habitats, while between 77% and 88% of bare soils were eventually planted with oil palm (see Supplementary Material). If the forest conversion statistics are modified to reflect the proportion of bare soils that originated from all types of forest habitats (disturbed and undisturbed, plus upland and wetland) and the proportion of bare soils that were allocated to oil palm plantations in the same temporal period, then approximately 48% (471,000 ha) of all plantations would have originated due to forest conversion. Moreover, if the area classified as bare soil within the peat polygon is included, then the area of oil palm plantations operating on peat soils in Sarawak would be approximately 476,000 ha (41% of all oil palm plantations in the state), which represents about 36% of all peat soils (~1.3 Mha) in the state (see Supplementary Material).

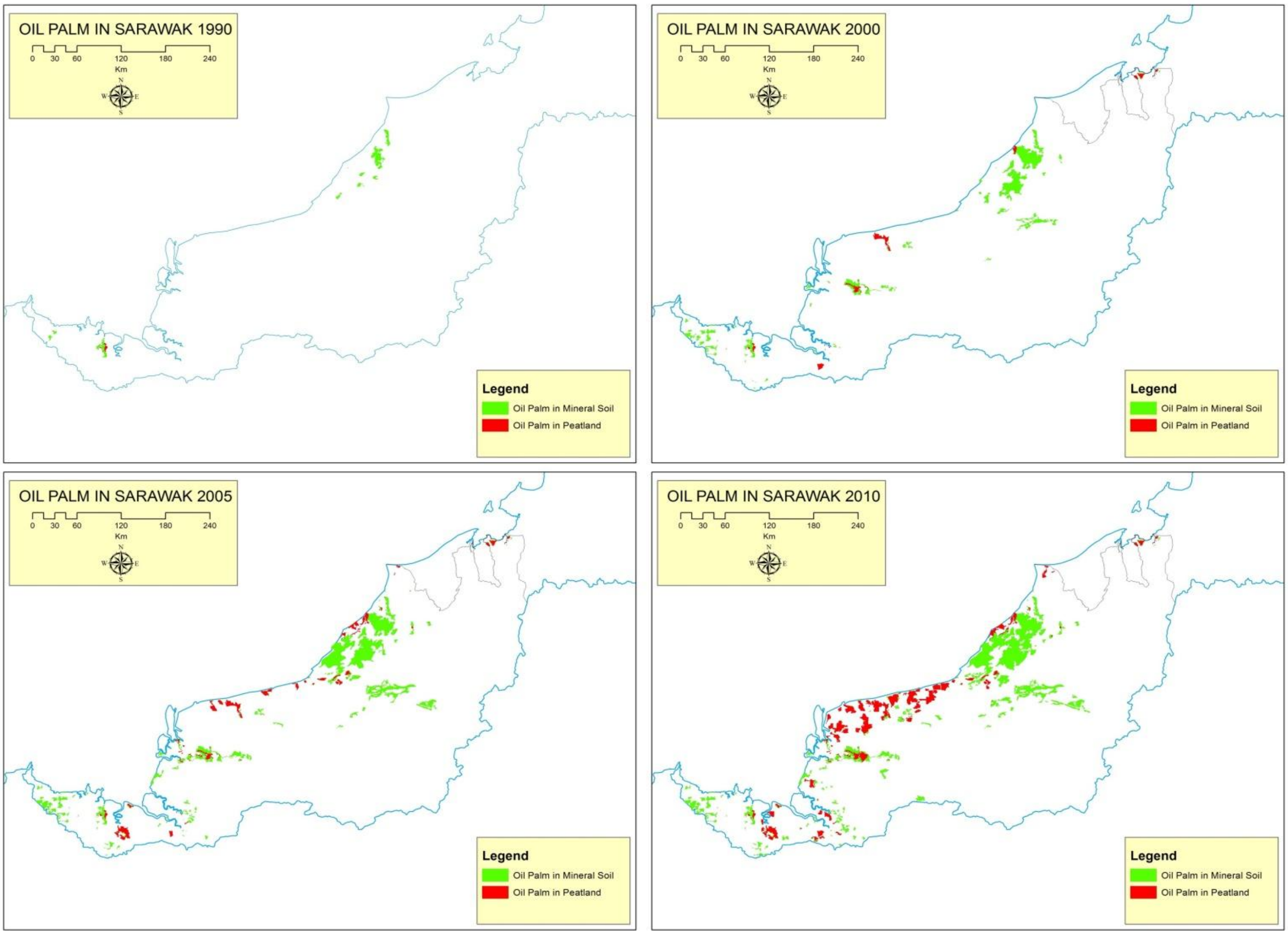

Figure 18. The expansion of oil palm plantations in Sarawak between 1990 and 2010.

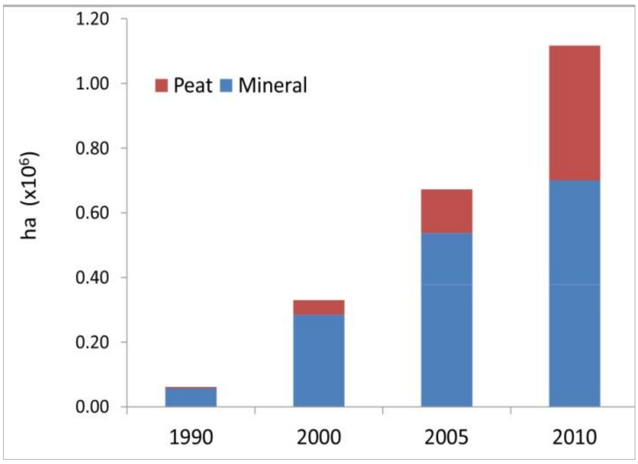

Figure 19. The area planted to oil palm in 1990, 2000, 2005 and 2010 on mineral and peat soil in Sarawak.

Table 20. Land cover area (10^3 ha) in 2000, 2005 and 2010 in Sarawak.

Aggregate Class	2000	2005	2010
Undisturbed Upland Forest	23	23	23
Disturbed Upland Forest	9,696	9,491	9,281
Upland Shrub and Grassland	6.2	27	34
Undisturbed Swamp Forest	1.7	1.4	1.4
Disturbed Swamp Forest	610	520	368
Swamp Shrub and Grassland	3.9	1.6	1.5
Agroforest and Plantation	1,263	1,180	1,007
Oil Palm Plantations	330	658	1,033
Intensive Agriculture	76	79	79
Bare soil	148	181	334
Others	290	285	286
Total	**12,448**	**12,448**	**12,448**

Table 21. Prior land use of new oil palm plantations established in Sarawak between 1990 and 2010.

Aggregate Class	1990 - 2000		2001 - 2005		2006 - 2010		1990 - 2010	
	10^3 ha	%	10^3 ha	%	10^3 ha	%	10^3 ha	%
Undisturbed Upland Forest	0.0	0.0	-	0.0	-	0.0	0	0.0
Disturbed Upland Forest	135	50	100	30.0	78	20	312	31.8
Upland Shrub & Grasslands	0.7	0.3	-	0.0	0.3	0.1	1.0	0.1
Undisturbed Swamp Forest	0.5	0.2	-	0.0	-	0	0.5	0.1
Disturbed Swamp Forest	21	7.8	20	6.2	43	11.3	84	8.6
Swamp Shrub & Grasslands	-	0	-	0.0	-	0.0	-	0.0
Agroforest & Plantation	87	33	129	38.9	120	31.4	336	34.3
Intensive Agriculture	-	0	2	0.6	-	0.0	2.1	0.2
Bare Soil	23	9	80	24.2	140	36.8	243	24.8
Others	0.4	0.2	0.7	0.2	0.0	0.0	1.2	0.1
Total New Plantations	**267**		**333**		**381**		**981**	

Papua New Guinea

The expansion of oil palm plantations in Papua New Guinea is similar to that documented for the Papua region of Indonesia. Large-scale plantations were established in the 1960s, largely on the island of New Britain, and by 2010 the country had a total of 133,516 ha (Figure 20 and Table 22). Growth in plantation area has fluctuated between 3 to 6% annually (2,440 to 4,261 ha) over the three temporal periods. The largest source of land cover type for this expansion has been disturbed upland forest (Table 23). Peats swamps are largely absent and no oil palm plantations were documented for that soil type in Papua New Guinea (Figure 20 and 21).

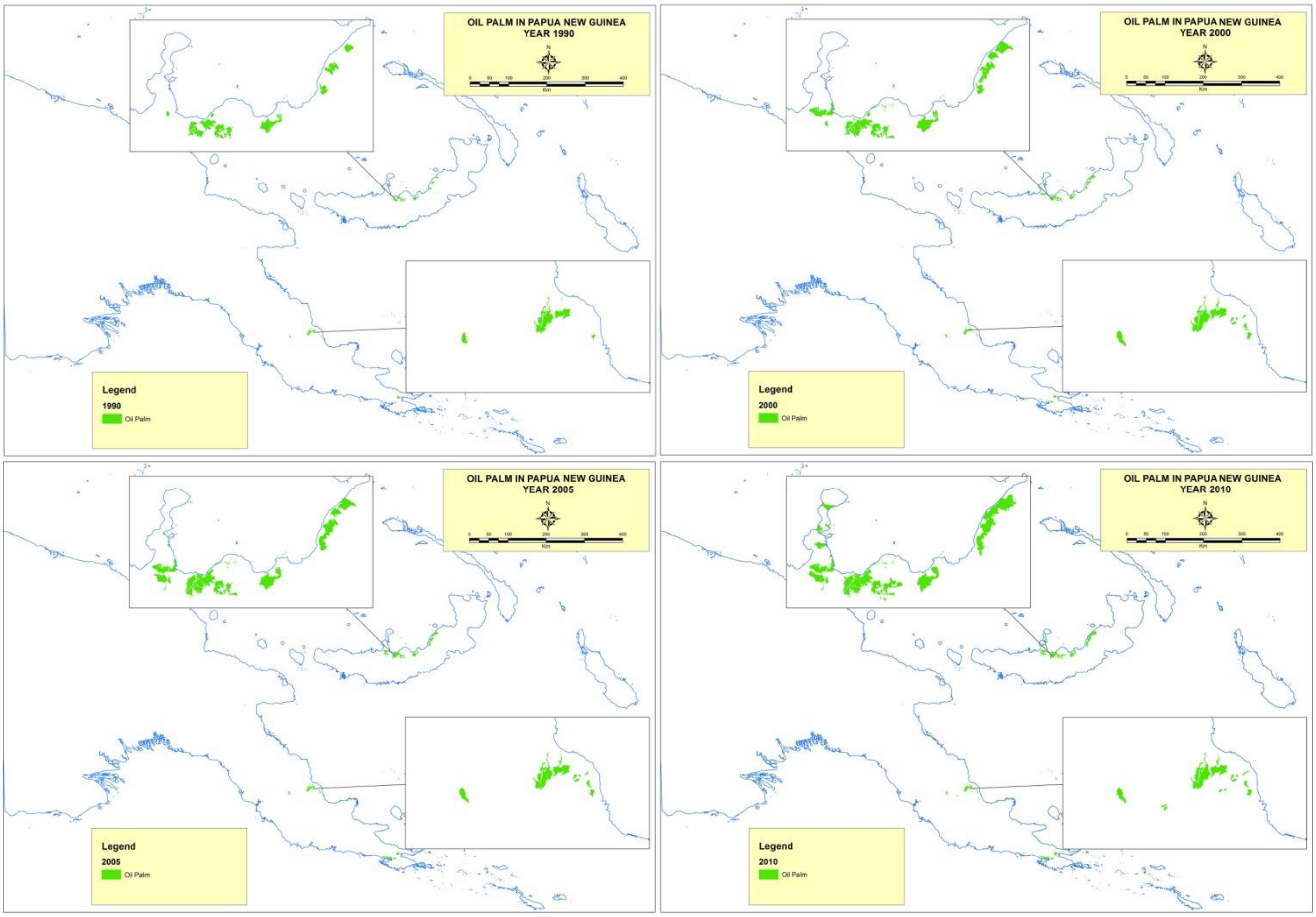

Figure 20. The expansion of oil palm plantations in Papua New Guinea between 1990 and 2010.

Table 22. Land cover area (10^3 ha) in 1990, 2000 and 2010 in Papua New Guinea.

Aggregate Class	1990	2,000	2010
Undisturbed Upland Forest	24,618	23,452	22,678
Disturbed Upland Forest	6,879	7,588	7,589
Upland Shrub and Grassland	5,307	5,680	6,292
Undisturbed Swamp Forest	1,538	1,533	1,521
Disturbed Swamp Forest	2,798	2,791	2,786
Swamp Shrub and Grassland	4,047	4,059	4,074
Agroforest and Plantation	302	302	304
Oil Palm Plantations	57	91	134
Intensive Agriculture	152	201	325
Bare Soil	81	77	73
Settlements, Mines and clouds	295	299	317
Mangroves	986	986	985
Water	289	289	273
Total	**47,349**	**47,349**	**47,349**

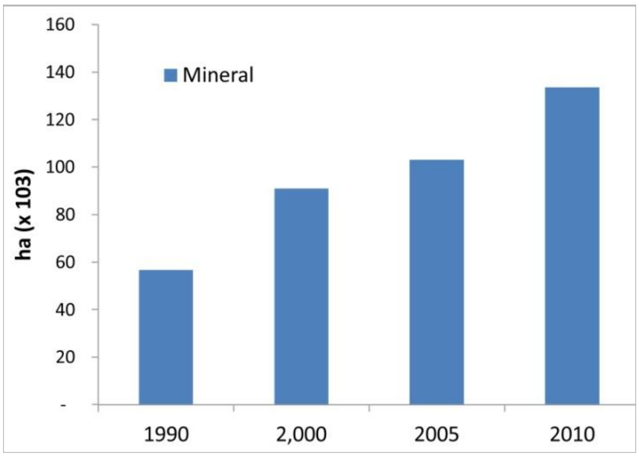

Figure 21. The area planted to oil palm in 1990, 2000, and 2010 on mineral soil in Papua New Guinea.

Table 23. Prior land use of new oil palm plantations established in Papua New Guinea between 1990 and 2010.

	1990 - 2000		2001 - 2005		2006 - 2010	
	ha	%	ha	%	ha	%
Undisturbed Upland Forest	4,302	12.6	316	0.7	4,618	6.0
Disturbed Upland Forest	11,854	34.6	25,045	58.8	36,899	48.0
Upland Shrub & Grasslands	16,123	47.0	11,789	27.7	27,912	36.3
Undisturbed Swamp Forest						
Disturbed Swamp Forest			169	0.4	169	0.2
Swamp Shrub & Grasslands			3,320	7.8	3,320	4.3
Agroforest & Plantation						
Intensive Agriculture	1,995	5.8	1,968	4.6	3,963	5.2
Bare Soil						
Others						
Total New Plantations	**34,274**		**42,607**		**76,881**	

DISCUSSION

Southeast Asia has the highest relative rate of deforestation in the humid tropics (Achard *et al.,* 2004; Sodhi *et.al*., 2005; Hansen, *et al.,* 2009; Houghton *et al.,* 2012) and the rapid development of the palm oil sector in Indonesia and Malaysia has contributed to this phenomenon. The expansion of oil palm plantations in Malaysia and Indonesia is one of several drivers of deforestation, however, and it is a misconception to allege that all oil palm plantations originate from forest conversion. This was recognized by Koh and Wilcove (2008) who estimated that between 1990 and 2005 between 55 to 59% of oil palm expansion in Malaysia and at least 56% in Indonesia were established as a direct result of forest conversion. That study did not differentiate between undisturbed and disturbed forests, although the authors did recognize that forest landscapes are often degraded by intensive logging and wildfire prior to their conversion to oil palm plantations. In a more comprehensive study (Wicke *et al*., 2011), the palm oil sector was identified as a major driver of forest cover loss in Sumatra and Kalimantan; these authors similarly recognized the complex nature of land cover change and the role of the forest sector as part of that dynamic. In both cases, the results and conclusions were limited by a reliance on secondary data derived largely from ministerial and sector reports (e.g., FAO, 2006; FAOSTAT, 2008).

Our study is based on a direct interpretation of satellite imagery for the entire region and shows that for the period between 1990 and 2010 approximately 36.5% of all oil plantations were established directly on some type of forest landscape, including both undisturbed and disturbed forest from both upland and swamp habitats (Table 5). If shrub land habitats are also included, and we assume that many of these are essentially highly degraded forest landscapes, then our results approach those reported by Koh and Wilcove (2008).

The distinction between primary and degraded or secondary forest has been one point of confusion when understanding the role of forest conversion in oil palm development. For example, the palm oil sector has made a point of emphasizing that they do not clear primary forests to establish plantations, a point which is essentially validated by our results. Nonetheless, disturbed forests also have biodiversity value (Hammer *et al.*, 2003; Peh *et al.*, 2005, 2006; Edwards *et al.,* 2010) and maintain significant carbon stocks (Pinard & Putz, 1996; Putz *et al.*, 2012) and this has motivated some authors to use terminology such as "primarily intact forests" (Carlson *et al.,* 2012b) or the oxymoronic "primary degraded forests" (Margono, *et al,* 2012). The definition of what constitutes "degraded" varies widely among authors, but in Indonesia it is assumed that areas classified as degraded land are a direct consequence of forest degradation (Wicke *et al.*, 2011; Margono, *et al.,* 2012). To avoid this type of terminological confusion, we delineate different types of land cover classes based on a combination of vegetation structure, degree of disturbance, and drainage (see Table 1 and 2); this allows us to document and track the transition between these categories so as to facilitate comparison and foster effective communication (Table 5 and Figure 10).

Our results also highlight the temporal and geographic variability associated with land use change and the oil palm sector. Forest conversion was much more important as a source of land for plantation expansion in the first and third temporal periods, but was less important between 2001 and 2005 when the sector converted an approximately equivalent area from rubber plantations and agroforest (Table 5). This trend was particularly notable in Sumatra where 85% of all new plantations established during the second temporal period occurred on existing "production" landscapes (Table 9) and in Peninsular Malaysia, where the conversion of plantations and agroforest landscapes over all three periods averaged 44% (Table 17). By contrast, other regions consistently converted large areas of forest landscapes to oil palm across all periods, particularly in Kalimantan, Sabah and Sarawak (Tables 11, 19 and 21).

Our results also show that the relatively low biomass landscapes that are converted to oil palm are themselves the consequence of forest degradation and conversion due to logging practices that are often compounded by the impact of wildfire. This dynamic is best described as a land use trajectory, and other studies have documented the impact of logging on forest cover prior to land clearing (Hansen *et al*., 2009; Margono *et al*., 2012). In Kalimantan, the transitional category is shrub land (see Figure10), while in Sumatra it was identified as mixed tree crops, which is a synonym for agroforest. Oil palm plantations have been established at multiple points along this trajectory.

Whether forest clearing is attributed to the oil palm sector or to the forest sector is partially dependent upon the time frame of the analysis; for example, if the analysis spans 10 years or more, the tendency is to allocate forest loss to the oil palm sector rather than to logging and fire. The impact of fire has been particularly large in Kalimantan, as evidenced by the conversion of degraded forest to shrub land between 1990 and 2000 linked to extensive and severe fires that occurred during the *El Niño* event of 97/98 (Hansen *et al.,* 2009). Our data show that this dynamic of forest degradation also occurred in the second temporal period, when approximately 50% of all forest loss occurred because areas classified as disturbed forest in 2000 were recognized as shrub land in 2005 (see Figure 10 and Supplementary Material). In Malaysia, the direct conversion of forest to oil palm was more common, particularly in Sabah and Sarawak (Tables 19 and 21), but the conversion of other land cover types, such as rubber plantations, was more important in Peninsular Malaysia (Table 17).

Comparison with other remote sensing studies

Due to the impact of deforestation and its threat to biodiversity conservation and climate change, land use change has been the focus of numerous studies in Southeast Asia (Stibig & Malingreau 2003; Miettinen *et al.,* 2012a, Hansen *et al.,* 2009, Broich *et al,.* 2011, Ekadinata & Dewi, 2011; Margono *et al.,* 2012). Our principal objective was to evaluate land use change linked to the expansion of oil palm plantations, but our results can also be used to estimate overall levels of deforestation (Table 24). Our results are both similar and distinct from other studies, an outcome that is to be expected when using different types of remote sensing data, classification methodologies, and definitional criteria when evaluating change on complex landscape mosaics (see Supplementary Material).

For example, our results differ significantly from a study that relied on moderate resolution MODIS images that compared two land cover maps for 2000 and 2010 covering Indonesia, Malaysia and Brunei (Miettinen *et al.,* 2012a). Their estimates of forest cover loss are 10% to 16% greater for Borneo and the Malay Peninsula, but are about 64% and 66% greater for Sumatra and Papua. A visual comparison of the maps shows that the greatest source of variance can be attributed to the nature of the output from an automatic pixel-based classification methodology when compared to an on-screen visual interpretation procedure. The automatic procedure identifies tens of thousands of small to medium patches of forest loss that are scattered across an otherwise intact forest matrix. In contrast, our visual interpretation grouped both types of pixels into a broad category defined as disturbed forest. The automatic procedure is efficient and objective when considering a limited number (e.g. 5) of land cover strata, but is impractical for developing a land cover classification with multiple types (e.g., 22).

Similar differences in data sources and classification methodologies likewise explain the differences between our study and a recent analysis based on high resolution radar images for Sarawak (SarVision, 2011). As in the MODIS-based study, an automated classification procedure produced an impressively accurate and precise high resolution map of forest cover that identifies the loss of tens of thousands of small deforestation patches, as well as hundreds of remnant forest patches that persist on agroforest landscapes. However, that study treats all forest pixels as equal entities, including those that are located in highly impacted landscapes with numerous logging roads and those located in protected areas with no visible disturbance: thus being precise in terms of forest change, but not necessarily accurate with respect to forest degradation. In contrast, we accurately, but imprecisely, lump these landscapes into categories of disturbed forest, which we assume has a lower carbon stock value than undisturbed forest (see Agus *et al.*, 2013a, 2013b – this publication). The combination of both pixel based methodologies and on-screen manual interpretation can provide both an accurate and precise estimate of disturbed and undisturbed forest cover types (Margono *et al.,* 2012).

Two studies used a combination of moderate resolution satellite imagery (MODIS) and Landsat images to track annual forest cover change in Indonesia between 2000 to 2005 (Hansen *et al.,* 2009) and between 2000 and 2008 (Boisch *et al.,* 2011). The combination of satellite imagery allowed the authors to take advantage of the high frequency of MODIS images and the higher resolution of Landsat imagery to produce estimates of forest conversion of greater temporal and spatial resolution. Nonetheless, these studies recognize only forest and non forest classes and lack the detail provided by multiclass land cover stratifications.

Table 24. Estimates of all types of deforestation in Indonesia and Malaysia based on a summation of the loss of all types of undisturbed and disturbed forest categories, including upland, swamp and mangrove habitats, either by conversion to some form of agriculture or plantation forestry or by the degradation of forest categories to scrub or grassland in both upland and swamp land cover types.

All types of forest (UDF+DIF+USF+DSF+UMF+DMF)	2000 – 2005				2005 – 2010			
	annual rate of deforestation		OP LUC as % total deforestation	% new OP	annual rate of deforestation		OP LUC as % total deforestation	% new OP
	10^3ha yr^{-1}	%			10^3ha yr^{-1}	%		
Indonesia*	**454**	**0.53**	**7.7**	**12**	**712**	**0.85**	**27**	**37**
Sumatra	*152*	*0.96*	*9.2*	*10*	*207*	*1.37*	*15*	*20*
Kalimantan	*225*	*0.66*	*8.9*	*28*	*454*	*1.37*	*35*	*44*
Papua	*85*	*0.25*	*1.2*	*23*	*42*	*0.12*	*5.6*	*79*
Malaysia	**159**	**0.71**	**60**	**34**	**142**	**0.66**	**68**	**35**
Peninsular	*57*	*0.89*	*56*	*28*	*20*	*0.33*	*56*	*12*
Sabah	*42*	*0.76*	*70*	*38*	*49*	*0.93*	*66*	*77*
Sarawak	*50*	*0.57*	*56*	*38*	*73*	*0.72*	*74*	*41*

Table 25. Comparisons of forest cover loss from three different studies using different satellite imagery, classification methodologies and temporal time periods for Sumatra and Kalimantan; values in parenthesis indicate increases in cover for that category.

Forest Cover	Land Cover (ha x10^6)					Annual Rates of Change (ha x10^3)				
	1990	2000	2005	2008	2010	1990 - 2000	2001 - 2005	2006 - 2010	2000 - 2008	2000 - 2010
Sumatra + Kalimantan (Hansen)[1]	68.9	55.7	52.7			1,320	600			
Sumatra + Kalimantan (Broich)[2]		57.8		57.8					653	
Sumatra + Kalimantan (Gunarso)[3]		50.1	48.2		44.9		377	668*		523
Sumatra (Margono)[4]		15.7			13.6					211
Sumatra (Gunarso)		15.9	15.1		14.1		152	206		179
Other tree-dominated types (Gunarso)										
Sumatra Agroforest[5]		2.9	2.2		2.1		141	13		77
Sumatra Shrub[6]		6.2	6.2		6.3		(5)	(21)		(13)
Kalimantan Agroforest		0.6	0.7		0.4		(14)	65*		25
Kalimantan Shrub		13.3	13.5		13.2		(48)	76*		14

[1]Hansen *et al., Environmental Research Letters,* 4, 034001 (2009)
[2]Broich *et al., Environmental Research Letters,* 6, 014010 (2011)
[3]Gunarso et al., (2013)- this publication, includes all forest classes: UDF, DIF, USF, DSF, UMF, DMF
[4]Margono *et al., Environmental Research Letters*, 7, 034010(2012)
[5]Includes MTC class only
[6]Includes SCH and SSH
* a mosaic images from 2009 and 2010

In addition, different definitional criteria may have caused them to incorporate agroforest areas into their forest class; for example, our decision to classify highly degraded forests as shrub land may overlap with their definition of forest. Not surprisingly, the differences among the three studies are less evident when other tree based systems (all types of forest, shrub land and agroforest) are aggregated in the results (Table 25).

Table 26. A comparison of two studies in Kalimantan between 2000 and 2009/2010 based on similar data, somewhat different classification methodologies and distinct classification criteria.

Carlson *et al.* (2012)		This study	
Land cover types	Source of OP plantations (%)	Land cover types	Source of OP plantations (%)
Primarily Intact Forest	47	Undisturbed Upland Forest	0.09
Logged Forest	22	Disturbed Upland Forest	35.1
Agroforest	21	Upland Shrub Land	38.8
Non Forest	10	Upland Grasslands	1.1
		Undisturbed Swamp Forest	0.1
		Disturbed Swamp Forest	8.4
		Swamp Shrub Land	7.6
		Swamp Grassland	0.1
		Rubber Plantations	1.3
		Pulp Plantations	0.3
		Mixed Tree Crops	0.3
		Rice Paddy Agriculture	0.01
		Upland Agriculture	4.1

A more recent study documented the extent and rate of oil palm expansion in Kalimantan between 1990 and 2010 (Carlson *et al.,* 2012b). That study documented approximately 3.1 Mha of oil palm plantations in the study area, a value slightly greater than the 2.9 Mha documented by our results. In both cases, the spatial area occupied by oil palm plantations was digitized manually on-screen and the difference between the two values may be the result of the use of several satellite images from 2009 in our study (vs. 2010) and the documented rate of change in Kalimantan of approximately 360,000 ha yr^{-1}, which would account for the difference between the two statistics. Other differences between the two studies were: 1) the use of an automated classification and change detection procedure to create four land cover types/change categories compared to our visual recognition of 22 land cover types, and 2) different definitional criteria for stratifying disturbed and undisturbed forest, versus primarily intact and logged forest (Table 26). Moreover, these authors classified only landscapes that fell within the polygons identified as oil palm plantations in 2010, which can be interpreted as the "plantation frontier" while we conducted a wall-to-wall classification for all of Kalimantan, which included the plantation frontier, as well as other areas that had been impacted by logging and fire, but have not (yet) been targeted for plantation development. Finally, Carlson *et al.* (2012b) applied the rates and sources of land cover change documented for the period between 1990 and 2010 to the period between 2000 and 2010, and assumed that the patterns of land use change in the first period would be the same as in the second period. In contrast, we documented the sources of land cover and rates of change for oil palm plantations separately for the periods: 1990 – 2000, 2001 – 2005 and 2006 – 2009/2010.

At first glance, results for the two studies are markedly dissimilar. Much of that difference, however, can be attributed to the use of different definitions and criteria for stratifying land cover classes (Table 26), particularly our decision to recognize a distinct tree-based, non forest "shrub land" category. Although almost 1.1 Mha of this category was converted into oil palm (see Table 11), it was simultaneously replenished by the ongoing degradation of disturbed forest (Figure 10), which we assume was due to the ongoing degradation caused by unsustainable logging practices and wildfire. Although we did not document the change between 1990 and 2000, massive wildfires caused the conversion of between 4.5 – 9 Mha of forests during the drought of the extreme *El Niño* event of 1997/98 (UNCHS, 2000; Hansen *et al*., 2009; van der Werff *et al.,* 2010). This phenomenon continues, as documented by our wall-to-wall study of land cover change in Kalimantan (Figure 10). Apparently, Carlson *et al.* (2012b) did not track land cover change between primarily intact or logged forest to either agroforest or non-forest, and assumed that change from any forest category to oil palm plantation was direct and did not include transitional degradation as a form of land cover change. The assumption by Carlson *et al.* (2012b) that the sources of land cover types for conversion to oil palm plantations in the 1990s would be the same in the next decade are not supported by our results (see Table 11), which show that relative proportion of forest conversion declined between 2001 and 2005, to then increase again in the last temporal period. A similar trend was documented by Hansen *et al.* (2009) who tracked annual changes in deforestation using MODIS images.

Differences in temporal periods and classification criteria limited our ability to compare the results of the Landsat based study for Malaysia (Rashid *et al.,* 2013 – this publication). Nonetheless, the results from the two studies broadly conform when evaluated for the extent and distribution of oil palm plantations, including those on peat soils, particularly if a significant portion of the category bare soil is assumed to be destined as oil palm plantations. However, there was less agreement concerning the land cover types that were the source of new oil palm plantations, in part because of less stratification in the data set (e.g., an "other" category that included at least 10 of the categories detailed in the Indonesian land cover classification). There were also discrepancies regarding the conversion of forest and rubber plantations; in the case of the former, the data set compiled by the Forest Research Institute of Malaysia showed increases in forest area between temporal periods.

Drivers of Deforestation

The differences in methodological approaches, including the use of different temporal periods, land cover definitions, and classification protocols, impacts on how the causes of deforestation are characterized and, consequently, attributed to different economic sectors. The definition of what constitutes a forest is precisely defined by foresters (FAO, 2007), but delineating forest cover from satellite images incorporates an element of subjectivity, particularly when visual techniques are employed, but also when pixel-based procedures use predefined cut-off points based on spectral indices. A large part of the differences among the various studies can be explained by differing definitions of forest and, more importantly when it comes to calculating GHG emissions (see Agus *et al*., 2013 – this publication), how to stratify the forest into different levels of disturbance. The potential for error is greatest on dynamic landscapes characterized by intermediate or even overlapping land cover types.

The drivers of land cover change are also usually not independent of one another. For example, timber exploitation almost always precedes plantation establishment and, in some cases, the two may be linked, as with wood salvage operations carried out as part of the land clearing process. In other cases, demonstrating a causal linkage is difficult, particularly if timber exploitation and plantation establishment are separated by several years or longer. In some regions, oil palm concessions have been used to fraudulently exploit timber resources with no intention of developing them as oil palm plantations (Sandker *et al.,* 2007). The impact of fire must also be considered, especially if it is sufficiently intense to create a tipping point that shifts a land cover from forest to a non forest over a period of a few weeks. Logging creates the conditions for fire by increasing forest litter and necromass, as well as opening the forest canopy to allow increased solar radiation to reach the forest floor and desiccate combustible material. Wildfires during periodic droughts can spread across large areas and have been particularly damaging to peat swamps where soil fires can damage root systems. Fire has been traditionally used to facilitate the development of oil palm plantations and carelessness may lead to uncontrolled fires that impact neighboring forest landscapes and cause them to shift from continuous forest to shrub land or agroforest.

The challenges linked to documenting land use change on highly dynamic landscapes can be managed by using short temporal periods to track change and by combining automatic pixel-based classification methodologies with visual interpretation to identify the economic and social actors that drive land use change (Margono *et al.,* 2012). In the specific case of Indonesia, our results show that there are multiple drivers of deforestation and that selection of temporal periods and the definitions of the parameters that define a forest can influence the allocation of deforestation to different economic sectors.

Oil palm plantations on peat in Malaysia and Indonesia

A total of approximately 2.43 Mha of oil palm plantations were established on peat soils in Indonesia and Malaysia by 2009/2010; this represents more than

9% of the total area of peat soils in these two countries if Papua is included, but almost 15% of the total area of peat in Peninsular Malaysia, Borneo and Sumatra. Sumatra leads in absolute areas of converted peat (Figure 7) and has converted approximately 19% (1.4 Mha) of its total peat area to oil palm plantations. The island also has large plantation areas dedicated to the cultivation of timber and cellulose, most of which is likewise planted on peat soils (Miettinen *et al.,* 2012c). Sarawak follows in absolute area with about 330,000 ha of oil palm plantation on peat in 2010 (25% of the total peat swamp area). However, if bare soils are included within this statistic, and in the case of Sarawak these are largely early stage oil palm plantations (88% between 2005 and 2010), then the total area of oil palm on peat in Sarawak surpasses 417,000 ha (37% of the total peat swamp area). The rate of change in the last temporal period of swamp forest in Sarawak was approximately 7% annually (59,620 ha) and nearly all of the loss of peat forest can be directly attributed to establishment of new oil palm plantations (see Supplementary Material).

Table 27. Comparison of three studies focusing on oil palm plantations on peat (10^6 ha).

	This study	Omar *et al.* (2010)	Miettinen *et al.* (2012)
Total Peat Area			
Malaysia	2.15	2.43	2.49
Peninsular	*0.72*	*0.72*	*0.85*
Sabah	*0.12*	*0.12*	*0.19*
Sarawak	*1.31*	*1.59*	*1.44*
Indonesia (excluding Papua)	13.04		13.00
Sumatra	*7.21*		*7.23*
Kalimantan	*5.83*		*5.77*
Total	**15.19**		**15.49**
Oil palm in 2010			
Malaysia	0.72	0.76	0.84
Peninsular	*0.21*	*0.30*	*0.26*
Sabah	*0.03*	*0.02*	*0.05*
Sarawak (including bare soil)	*0.48*	*0.44*	*0.53*
Indonesia (excluding Papua)	1.71		1.29
Sumatra	*1.40*		*1.03*
Kalimantan	*0.31*		*0.26*
Total	**2.42**		**2.13**

In contrast, in neighboring Kalimantan large areas of peat have been degraded and abandoned without any productive use or effort to restore their ecological functionality (Figure 10). Our results documenting the extent of oil palm plantations are similar to two other studies that used high resolution SPOT images (Table 27). All relied on soil maps to delineate the spatial extent of peat swamps and the extent of oil palm plantations were all derived by a manual on-screen digitizing methodology. The differences among the studies are most probably due to the spatial area defined by different peat soil polygons.

CONCLUSIONS

The historical trend in oil palm plantation development in the region has stayed remarkably steady between 7 and 7.7% annual growth rate over twenty years. There have been short term variations and we document one of these in the second temporal period when there was a tendency to convert previously cleared lands and other forms of plantations to oil palm. Similarly, there are measurable differences among the various sub national units: Sumatra, Peninsular Malaysia, and Sabah all showing rates that have decreased considerably in the last temporal period. The absolute area of new plantations in Sumatra remains large, but the annual rate of growth has declined from 7.6% initially to 3.8% in the last five year period. Even in Sarawak, which had annual growth rates between 15 and 20% between 1990 and 2005, growth has slowed somewhat, although there is no indication that the rates of change on peat soils is decreasing. Kalimantan continues to expand at near exponential rates of growth, a trend that we believe will moderate in the near future; as in other regions, the conversion of peat soils in Kalimantan has increased over time. If past history is a reliable guide and demand for palm oil continues to grow, it is likely that expansion will continue at 7% annual rates over the short term, however future expansion might shift to the frontier landscapes of Papua and Papua New Guinea.

The production of palm oil is only one driver of deforestation. In Indonesia, the largest single cause of historical forest loss is probably due to intensive logging and the impact of fire, which in combination have led to the progressive degradation of large areas of forest landscapes into agroforest or shrub land. In Malaysia, the direct conversion of forest to oil palm was more common, particularly in Sabah and Sarawak, but the conversion of other types of land use, such as rubber was more important in Peninsular Malaysia.

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HISTORICAL CO_2 EMISSIONS FROM LAND USE AND LAND USE CHANGE FROM THE OIL PALM INDUSTRY IN INDONESIA, MALAYSIA AND PAPUA NEW GUINEA

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ABSTRACT

The CO_2 emissions from land use change (LUC), peat fires and peat oxidation due to the establishment and operations of industrial oil palm plantations were estimated for the major palm oil producing regions of Indonesia (Sumatra, Kalimantan and Papua), Malaysia (Peninsular Malaysia, Sarawak and Sabah) and Papua New Guinea. Measurements of oil palm expansion were based on the visual interpretation of Landsat images from 1990, 2000, 2005, and 2009/2010 that produced a 22 x 22 LUC matrix, which was used in conjunction with emission factors calculated from the differences in the mean value of published reports for above ground carbon (AGC) for each land cover class (e.g., 189 Mg C ha^{-1} for undisturbed forest, 104 Mg C ha^{-1} for disturbed forest, 30 Mg C ha^{-1} for shrub land, 36 Mg C ha^{-1} for oil palm plantations). The emission factor for peat oxidation for oil palm plantations operating on peat soils (43 Mg CO_2 ha^{-1} yr^{-1}) was based on a review of the scientific literature, while the emission factors for peat fires were based on the assumption that fires were used historically to clear land when establishing oil palm plantations in swamp forest (333 Mg CO_2 ha^{-1}) and swamp shrub land (110 Mg CO_2 ha^{-1}).

The total area of oil palm plantations increased from 3.5 to 13.1 Mha between 1990 and 2010 at a mean annual rate of approximately 7%. Over this 20 year period, the direct conversion of natural forest preceded the establishment of approximately 3.5 Mha (36.6%) of new oil palm plantations, with the remainder resulting from the conversion of moderate to low biomass vegetation types, including 1.7 Mha of shrub and grassland habitats (17.6%) and 3.5 Mha of land cover types (37.5%) that had been converted previously to field crops, agroforest or other types of plantations, and 0.9 Mha of other land cover categories (9.5%).

The net emissions of CO_2 from oil palm plantations in the study area resulting from changes in AGC due to LUC, peat fires and peat oxidation increased from 92 to 106 to 184 Tg CO_2 yr^{-1} between the first (1990 – 2000), second (2001 – 2005) and third (2006 – 2009/10) temporal periods. The proportion of CO_2 emissions that originated from AGC due to LUC decreased between the first and second temporal period, but increased in the third (55 to 42 to 67 Tg CO_2 yr^{-1}); the emissions from peat fires linked to LUC tracked those of AGC (12 to 8 to 29 Tg CO_2 yr^{-1}). In contrast, the emissions from the oxidation of peat from plantations operating on partially drained peat soils increased steadily over all three temporal periods (26 to 56 to 88 Tg CO_2 yr^{-1}). Emissions from AGC due to LUC and peat fires are one time emissions that occur at the time of plantation establishment, but peat oxidation results in long-term, annual recurring emissions. By 2010, plantations on peat constituted 18% (2.4 Mha) of the spatial footprint of palm oil, but emission from peat fires and peat oxidation were the source of approximately 64% (118 Tg CO_2 yr^{-1}) of the total emissions from land use linked to industrial scale oil palm plantations.

Finally, we compared the CO_2 emissions from oil palm with the emissions from AGC due to LUC and peat oxidation from other types of land use; emissions from peat fires were excluded due the lack of data on the incidence of fire in other land use categories. We estimate that oil palm was responsible for approximately 13% of the total of these two types of emissions between 2000 and 2005 and 18% between 2006 and 2009/2010, based on total estimated emissions of 698 and 792 Tg CO_2 yr^{-1}, respectively. The largest source of CO_2 emissions originated from a land use trajectory that caused undisturbed forest to be degraded to disturbed forest and then to shrub land, presumably the result of logging and wildfire. Emissions from AGC from this type of forest loss and degradation was estimated at 267 Tg CO_2 yr^{-1} between 2000 and 2005 (39% of the total) and 285 Tg CO_2 yr^{-1} between 2006 and 2009/2010 (36% of the total). The sources of uncertainty in this and other published studies are discussed and represent a potential range that is an order of magnitude smaller or greater than the modeled estimates presented in this study. Prioritizing the use of shrub and grassland on mineral soil and avoiding of the use of peat soils will reduce emission significantly, as will enforcing the ban on fire for land clearing.

Keywords: *Land use change, CO_2 emissions, peat oxidation, low-carbon shrubland rehabilitation*

INTRODUCTION

The palm oil industry has grown from providing less than 5% of the global supply of vegetable oils in 1970 to providing approximately 35% of the global market demand (Teoh, 2010). The rapid growth in the production of palm oil reflects the success of a highly efficient plantation system and the inherent productive capacity of the oil palm (*Elaeis guineensis*). The palm oil industry is expected to expand in the near to medium term in response to the demand for vegetable oil as food in emerging economies and developing countries, and potentially, as a biofuel feedstock in North America and Europe. The plantation model of production is widespread and has existed for more than a century in Africa, Latin America and Southeast Asia (Corley & Tinker, 2003), but it has reached its most sophisticated level of operation in Malaysia and Indonesia, which together produce approximately 85% of global supplies of palm oil. Indonesia is expected to expand the area under cultivation by about 50%, from approximately 8 million ha in 2010 to 12 Mha by 2020 (Teoh, 2010), while Malaysia is expected to increase its oil palm plantations by only 28% due to the limitation of available land resources (Dompok, 2011). Other areas, particularly Papua New Guinea, Thailand, West Africa and South America also are expected to increase oil palm plantations in response to the demand from world markets.

The rapid expansion of oil palm plantations has generated a heated debate about the environmental impacts of palm oil production, particularly as it relates to impacts on climate change, biodiversity and the use of pesticides; social conflicts associated with land disputes and the loss of access to forest resources by local communities have also generate controversy (Panapanaan *et al.,* 2009). The environmental disputes are linked to the widespread assumption that a large proportion of palm oil plantations have been created as a direct consequence of forest clearing. This assumption is challenged by the palm oil industry that asserts that most existing oil pam plantations have been established on lands that were degraded forest, shrub land and rubber plantations (Smith, 2011). Recent studies from Indonesia provide evidence that land cover is dynamic and complex. Deforestation has been associated with the expansion of plantation estates and cropland; however, agroforest landscapes where coffee, cacao, citrus and timber are grown as part of a diversified smallholder production system have decreased gradually since 1990 and so are also likely to be involved. Simultaneously, the loss of forest cover has been linked with the increase in shrub land between 1990 and 2000, presumably due to forest degradation, but this type of land cover decreased between 2000 and 2005, as it was converted to more productive types of land use including oil pam (Ekadinata & Dewi, 2011).

Several studies documenting deforestation have been completed for both Malaysia and Indonesia (Stibig & Malingrea, 2003; Hansen *et al.,* 2009; Miettinen *et al.,* 2011, 2012a) and both governments provide periodic reports to the global database on forest resources (FAO, 2010). However, detailed studies that quantify land use change (LUC) specific for the palm oil sector are nonexistent or incomplete. In Indonesia, Ekadinata & Dewi (2011) analyzed land cover changes for two temporal periods: 1990 – 2000 and 2001 – 2005, but treated all types of industrial plantations as a single category, including oil palm, pulp and paper and rubber. Similarly, the Indonesian Ministry of Forestry (MoF, 2008) analyzed land use change for 2000 – 2003 and 2004 – 2006 and likewise grouped all plantation types into a single category (see WRI, 2008). In Malaysia, a variety of government institutions have tracked forest cover and land use change and have provided detailed information on the expansion of oil palm plantations and changes in forest cover; unfortunately, those studies use different data sources and classification methodologies and lack consistency in the definition of forest between temporal benchmarks making the estimates of change between oil palm expansion and deforestation difficult to verify (Rashid *et al.,* 2013 – this publication). The most widely cited estimate of deforestation attributed to oil palm plantations is based on a reinterpretation of the national reports provided by government ministries to the Forest Resource Assessment program of the Food and Agriculture Organization (FAO, 2010) covering the period between 1990 and 2005. This information has been reinterpreted to provide an estimate that approximately 55 – 59% of oil palm expansion in Malaysia and Indonesia has occurred at the expense of forests (Koh & Wilcove, 2008). It is important to note, however, that this conclusion is based on secondary sources unverified by remote sensing studies, and the FAO database is not considered to be reliable for many tropical forest countries by some remote sensing scientists (Grainger, 2007; Olander *et al.,* 2008).

The controversies surrounding CO_2 emissions and land use are compounded by the uncertainty in the dimensions and variability of above and below ground carbon stocks in natural, degraded, and anthropogenic landscapes. This uncertainty is a function of the variability inherent in any natural ecosystem (Saatchi *et al.,* 2011) and the temporal changes that occur as one class transitions into another (Lambin *et al.,* 2003). Land use change may be abrupt in the case of the conversion of forest habitat to a plantation estate or gradual when primary forest is logged, logged again, and exposed to wildfire prior to its conversion to agriculture. Moreover, the identification of transitional categories is subject to the time span used for the study; for example, a temporal comparison spanning a decade or longer will often document a transition from undisturbed forest to plantation, but a multi-temporal study with shorter periods might reveal that undisturbed forest first become degraded forest and then shrub land, prior to its conversion to some form of productive activity. In addition, the selection of carbon stock values can greatly impact the estimates of net CO_2 emissions, particularly in light of the capacity for plantation landscapes to capture and store significant amounts of carbon (Wautersa *et al.*, 2008; Henson*,* 2009).

Another major controversy is related to the conversion of coastal peat swamps to plantation estates; this type of production strategy requires the partial drainage of these wetland habitats, which leads to the oxidation of peat and the emission of CO_2. Drainage and oxidation causes the peat soils to subside and reduces their capacity to regulate the surrounding hydrology; if the process continues, the underlying mineral soil layer will eventually become exposed or, more likely, the subsidence will approximate the level in adjacent coastal water bodies that are often chemically saline (Hooijer *et al.,* 2010). The dimensions of CO_2 emissions from drained and converted peat swamps are subject to numerous uncertainties and have been a source of contention over the last decade. Estimates of the emission from peat oxidation vary widely, ranging from a low of 26 Mg CO_2 ha^{-1} yr^{-1} (Jauhiainen *et al.,* 2001) in agricultural land to a high of 100 Mg CO_2 ha^{-1} yr^{-1} in oil palm plantations (Hooijer *et al.,* 2012; Page *et al.,* 2011). The uncertainty in these estimates is related to both the physical nature of tropical peat and a lack of studies that adequately address the natural sources of variability, as well as disagreements among soil scientists on how to directly measure CO_2 emissions and the components of a modelling approach that estimates emissions in the absence of direct measurements (Melling *et al.,* 2005; Hooijer *et al.,* 2010; 2012; Agus *et al.,* 2012).

This paper seeks to clarify some of the uncertainties outlined in the previous paragraphs and provide a more robust estimate of CO_2 emissions linked to land use change caused by the palm oil sector. To do this, we documented the full trajectory of the conversion of forest landscapes to oil palm plantations, as well as evaluating how other land cover types have contributed to the expansion of the oil palm plantations. Our primary goal is to provide an objective estimate of the CO_2 emissions from the establishment of new oil palm plantations and to model the emissions from plantations established on peat soils. As part of that process, we provide estimates of the greenhouse gas (GHG) emissions linked to other productive sectors and place the emissions directly linked to palm oil in the broader context of land cover and land use change.

INFORMATION SOURCES AND METHODOLOGIES

This paper represents a synthesis of information that comes largely from two different sources:

1) An original analysis of land cover and land cover change for two decades for the principal palm oil producing regions in Indonesia (Sumatra and Kalimantan) and Malaysia (Peninsular Malaysia, Sabah and Sarawak), as well as the regions most likely to be the focus for future palm oil expansion (Indonesian

Papua and Papua New Guinea)(Gunarso *et al.,* 2013 – this publication).

2) A review of the published literature of carbon stock values for above and below ground biomass for these same geographies and a critical evaluation of the range of values reported for CO_2 emission from peat and the underlying assumptions that are used when estimating them (Agus *et al.,* 2013 – this publication).

Land Cover and Land Use Change

The spatial extent and expansion of oil palm estates was documented for three temporal periods (1990 – 2000, 2001 – 2005 and 2006 – 2009/2010) based on a visual interpretation of Landsat satellite images (Gunarso *et al.,* 2013 – this publication). The land cover stratification is composed of 22 classes, which was based on a harmonization of two similar systems used by the Ministry of Forestry (21 classes) and the Ministry of the Agriculture (23 classes) of the Republic of Indonesia (Table 1). The same system was used for the Malaysian states and Papua New Guinea to ensure uniform criteria for all regions (see Table 1 – Gunarso *et al.,* 2013 – this publication). Experienced GIS technicians visually identified similar groups of pixels based on spectral attributes, geometric patterns, and landscape context to digitally trace polygons on the computer screen. Land use change between each of the different land cover categories was documented and summarized via a 22 x 22 land use change matrix for each temporal period and for each sub-region included in the study. The results were pooled using aggregate categories to facilitate the communication of the results (see first column in Table 1).

Table 1. Emission factors used for the calculation of emission for Indonesia, Malaysia and Papua New Guinea for the above ground (biomass) time average carbon stock and peat oxidation for land use on peat.

Land Cover				Time average above ground carbon stocks		Peat oxidation		Peat fire emissions from conversion
Aggregate	Code	Class	Description	Selected Value (Mg C ha^{-1})	Range (Mg C ha^{-1})	Water Table Depth (cm)	Peat (Mg CO_2 ha^{-1} yr^{-1})	(Mg CO_2 ha^{-1})
Natural Forest	UDF	Undisturbed Upland Forest	Natural forest cover with dense canopy (> 80%), no signs of logging roads; image with high NDVI and infrared channels, lower value in visible channels.	189	61 - 399	n.a.	n.a.	n.a.
	DIF	Disturbed Upland Forest	Natural forest with visible logging roads and clearings visible; image with lower NDVI and infrared channels	104	33 - 250	n.a.	n.a.	n.a.
Degraded Non Forest	SCH	Upland Shrub land	Woody vegetation usually less than 5 m in stature, often regeneration following swidden agriculture activities or intensive logging.	30	27 – 35	n.a.	n.a.	n.a.
	GRS	Upland Grassland	Extensive cover of grasses with scattered shrubs or trees.	3	2 – 4	n.a.	n.a.	n.a.

Table 1. Emission factors for the above ground (biomass) time average carbon stock and peat oxidation (continued).

Land cover				Time average above ground carbon stocks		Peat oxidation		Peat fire emissions from conversion
Aggregate	Code	Class	Description	Selected Value (Mg C ha^{-1})	Range (Mg C ha^{-1})	Water Table Depth (cm)	Peat (Mg CO_2 ha^{-1} yr^{-1})	(Mg CO_2 ha^{-1})
Swamp Forest	USF	Undisturbed Swamp Forest	Natural forest with temporary or permanent inundation.	162	90 – 200	0	0	330
	DSF	Disturbed Swamp Forest	Natural forest cover with indications of logging activity and influence of drainage	84	33 – 155	30	22	330
Open Swamp	SSH	Swamp Shrub land	Woody vegetation less than 5 m in stature, often regeneration following swidden agriculture or logging in areas, mostly affected by drainage	28	18 – 35	30	22	110
	SGR	Swamp Grassland	Extensive cover of grasses with scattered shrubs or trees in inundated area.	2	2	30	22	0
Agroforest & Plantations	TPL	Timber Plantation	Monoculture timber or pulp plantation; canopy cover between 30-50%.	44	29 – 70	50	36	0
	MTC	Mixed Tree Crops (Agroforest)	Agroforest with > 30% of tree cover; usually to settlements and roads; includes rubber, coffee, cacao and home garden.	54	30 – 77	50	36	0
	RPL	Rubber plantation	Traditional and monoculture rubber plantations, sometimes mixed with rubber agroforestry.	55	31 - 89	50	36	0
Oil Palm Plantation	OPL	Oil Palm Plantation	Large Scale Oil Palm Plantation.	36	22 – 60	60	43	0
Bare Soil	BRL	Bare Soil	Exposed soil, gravel, or sand; frequently associated with areas undergoing land use change	36[1]		0	0	0
Agriculture	DCL	Cultivation Land in Upland soils	Open area with herbaceous vegetation; sometimes mixed with shrub land; usually associated with settlements.	11	8 - 12.5	30	22	0
	RCF	Rice Field	Open, flat area subject to inundation; usually associated with settlement and irrigation structure.	2	2	10	7	0

[1]The value of 36 Mg ha^{-1} was used as default carbon stock value in order to avoid introducing artifacts into estimates of oil palm emissions

Table 1. Emission factors for the above ground (biomass) time average carbon stock and peat oxidation (continued).

Land cover				Time average above ground carbon stocks		Peat oxidation		Peat fire emissions from conversion
Aggregate	Code	Class	Description	Selected Value (Mg C ha^{-1})	Range (Mg C ha^{-1})	Water Table Depth (cm)	Peat (Mg CO_2 ha^{-1} yr^{-1})	(Mg CO_2 ha^{-1})
Other	SET	Settlements	Urban areas, towns and villages; associated with road network	7	4 - 10	70	50	0
	MIN	Mining	Open area with mining activities.	0		100	72	0
	UDM	Undisturbed Mangrove	Forest area along the coastline with high density of mangrove tree species; no evidence of logging.	148	85 - 200	n.a.	n.a.	n.a.
	DIM	Disturbed Mangrove	Natural forest along the coast with mangrove species, with evidence of logging.	101	77 - 120	n.a.	n.a.	n.a.
	CFP	Coastal Fish Pond	Open coastal area with block pattern and always inundated.			n.a.	n.a.	n.a.
	WAB	Water bodies	Water bodies; images with low reflectance in all bands.			n.a.	n.a.	n.a.
	NCL	Not Classified Cloud	High reflectance in all bands			n.a.	n.a.	0

Carbon Stocks and Emission Factors

Above ground carbon (AGC) can be either a source or sink of atmospheric CO_2 depending on the difference between the carbon stock of the land prior to and after land use change (LUC). The emission factors from changes in AGC due to LUC are the differences between the mean values of published reports of the carbon stocks for each of the 22 land cover types listed in Table 1 (see review by Agus *et al.,* 2013 - this publication). The variability in the above ground carbon of forest and shrub land vegetation types is due to the interactions of biodiversity and ecological processes, as well as human disturbance from logging and fire. In contrast, crop land and plantation estates are characterized by simple vegetation structure and uniform planting density. Nonetheless, published reports for the carbon stock of oil palm plantations vary by as much as 50%, because different studies include or exclude below ground biomass, ground vegetation, litter and persistent leaf bases that represent short-term carbon pools. The value of 36 Mg ha^{-1} adopted in this study is the mean of several studies that estimate the time-averaged carbon stock of an oil palm plantation that starts near zero to reach more than 155 Mg C ha^{-1} for a 25-yr old plantation (see Agus *et al.,* 2013 - this publication). In the case of bare soils, a transitional category of uncertain origin, we use a value of 36 Mg ha^{-1} as default carbon stock value in order to avoid introducing artifacts into estimates of oil palm emissions. Similarly, obvious errors in land cover classification that produced illogical land use change outcomes (e.g., apparent conversion of water bodies to oil palm) were excluded from the analysis.

The decomposition of peat, also known as peat oxidation, is the most important source of CO_2 emission in oil palm plantations operating on peat soils. Upon partial drainage and conversion, the functional attributes of peat soils change from being a net sink to become a net source of CO_2 (Hooijer *et al.* 2006; Agus & Subiksa 2008; Agus *et al.,* 2012). The rate of emission is primarily a function of the depth of drainage, but other factors such as local climate and peat maturity also influence the rate of decomposition. Estimates of CO_2 emissions from peat oxidation under different conditions remain uncertain, in part due the difficulty of distinguishing between the autotrophic respiration from roots and the heterotrophic respiration from the soil biota that mediates decomposition (see review by Agus *et al.,* 2013 – this publication). We used as a basis the emission factor of 0.91 Mg CO_2 $ha^{-1}cm^{-1}$ (Hooijer *et al*. 2010), but modified that value by a coefficient of 0.79 to correct for the root-related emission based on the studies by Jauhiainen *et al*. (2012). In our model, we assume that oil palm plantations on peat soils have a mean water table depth between 50 and 70 cm, which generates emission estimates between 36 to 50 Mg CO_2 ha^{-1} yr^{-1} with an average value of 43 Mg CO_2 ha^{-1} yr^{-1}.

Peat fires are another major source of CO_2 emissions linked to the cultivation of oil palm on peat. Although the use of fire is on the decline, it was a common management practice throughout the temporal periods described in this study (Schrier-Uijl *et al.,* 2013 – this publication). Peat soils must be drained prior to plantation establishment, but the depth of the water table and the degree of soil dryness varies widely across years: When peat soils are dry, they catch fire and burn. The depth of peat fires range from more than 50 cm during severe drought, such as the mega *El Niño* event of 1997/98 (Page *et al.* 2002), to zero during unusually wet years. We assume that when swamp forest is converted to oil palm, an average of 15 cm of peat is consumed by fire and, because fire is less intense when shrub land is cleared, an average of only 5 cm of peat is lost. In both cases, we assume that peat has a mean carbon content of 0.06 Mg m^{-3}. This combination of peat depth and carbon density were used to calculate an emission factor of 330 Mg CO_2 ha^{-1} for plantations established on forest landscapes and 110 Mg CO_2 ha^{-1} on shrub land (Agus *et al.,* 2012; 2013 – this publication). We assume that fire has not been used and there were no emissions when oil palm plantation were established on cropland, agroforest, other types of plantation or any of the miscellaneous land cover categories. In the case of bare soil, in those areas where this land cover class was documented as being an integral part of the oil palm land use dynamic (Peninsular Malaysia and Sarawak), we treated that proportion of bare soil area as oil palm plantations according to the relative area of bare soils that had been planted to oil palm in the previous temporal period.

Emission Calculation

The estimate of the net carbon emissions was based on IPCC (2006):

*Emission = Activity data*Emission factor*

Activity data is the area under specific land use or undergoing land use change (LUC) within a defined period of time. The Activity data is based on the 22 x 22 LUC matrix for each subregion for each period at national or sub-national level. Emission factor is the change in carbon stock in every major pool or emission rate in case of peat oxidation. The net emission can be calculated as:

$$E = E_a - S_a + E_{bo} + E_{pf}$$

where E is net CO_2 emission, E_a is emission from AGC due to LUC, S_a is sequestration of CO_2 from the atmosphere into crop biomass of the succeeding land uses, E_{bo} is emission from below ground soil organic matter decomposition (peat oxidation), and E_{pf} is emission due to peat fire.

Emissions from AGC due to LUC are calculated based on carbon stock change:

E_a – S_a = (Biomass C stock of the initial land use – Time-averaged plant biomass C stock in the successive land use) * 44/12 * A/t

Emissions from peat oxidation are estimated based on mean depth of drainage and observed rates of CO_2 emission corrected for root respiration:

E_{bo} = 0.91 *0.79 * drainage depth * A/t

Emissions from peat fires are based on carbon density, burn depth and the area of new planting:

E_{pf} = (C density * burn depth) * 44/12 * A/t

The coefficient 44/12 or 3.67 is the conversion factor from C to CO_2, based on atomic weights of C and O of 12 and 16, respectively, t is the period (number of years) of analysis and A is the activity data or area of land use. Quantitative information are expressed using the standard prefixes of the International System of Units (SI): a metric ton is Mg (g x10^6), a million metric tons is Tg (g x10^{12}) and a million hectares is Mha (ha x10^6).

RESULTS

Land Use Change

The total land surface dedicated to the cultivation of oil palm has increased dramatically in Southeast Asia expanding from 3.5 Mha in 1990 to more than 13.1 Mha in 2009/2010 (Table 2); much of that expansion has occurred at the expense of forest. When summed over all regions and for all three temporal periods, forest landscapes were the source of approximately 36.6% of all new oil palm plantations: 25.4% from upland forest and 11% from swamp forests, including both undisturbed and disturbed forest (see Gunarso *et al*., 2013 – this publication). The comparison of soil and land cover maps show the proportion of all oil palm plantations on peat soils at approximately 2.4 Mha in 2009/2010, representing about 18% of all plantations in the study area (see Gunarso *et al*., 2013 – this publication).

Table 2. Oil palm development in Indonesia (Sumatra, Kalimantan and Papua) and Malaysia on peatland and mineral soils (million hectares).

Country, soil	1990	2000	2005	2010
Indonesia	**1.34**	**3.68**	**5.16**	**7.72**
Peat	*0.27*	*0.72*	*1.05*	*1.70*
Mineral	*1.07*	*2.95*	*4.10*	*6.02*
Malaysia	**2.08**	**3.53**	**4.59**	**5.38**
Peat	*0.15*	*0.28*	*0.40*	*0.72*
Mineral	*1.93*	*3.25*	*4.19*	*4.66*
Papua New Guinea	**0.06**	**0.09**	**0.10**	**0.13**
Total	**3.47**	**7.29**	**9.85**	**13.23**

The mean rate of expansion has increased from approximately 373,000 in the 1990s to more than 735,000 ha yr^{-1} in the last temporal period, maintaining an annual growth of approximately 7% over two decades (Figure 1). The development and early expansion of the industry occurred first in Peninsular Malaysia and Sumatra prior to 1990, but expanded over the next two decades to include both the Indonesian and Malaysian regions on the island of Borneo (Gunarso *et al.,* 2013 – this publication). Growth in Malaysia has been more or less constant, but slowed slightly in the last temporal period, and shifted from Peninsular Malaysia to the states of Sabah and Sarawak over time. Indonesia surpassed Malaysia as the world's largest producer of palm oil in 2007 due largely to expansion in Sumatra; however, growth of new plantations in Kalimantan predominated between 2006 and 2009/2010. In the last five year period, expansion slowed in Peninsular Malaysia, Sabah, and Sumatra, but increased dramatically in Kalimantan, while holding steady in Sarawak, Papua and Papua New Guinea (Figure 1).

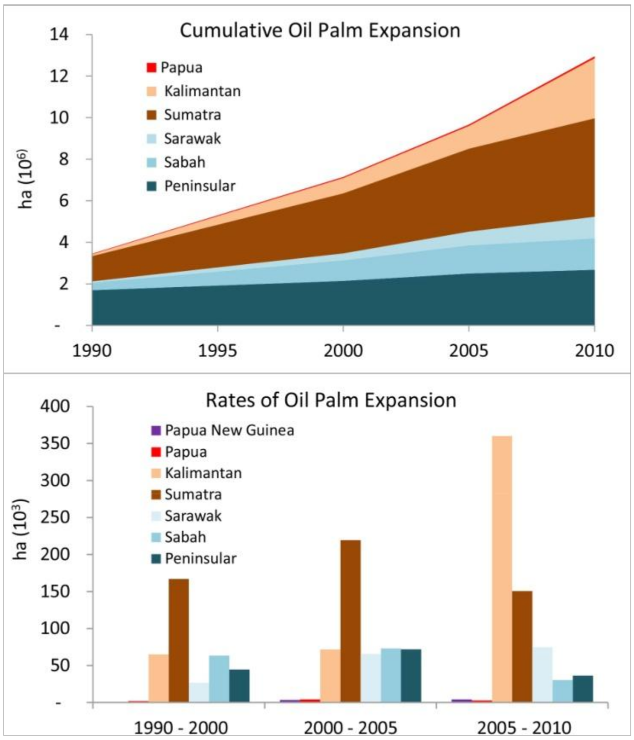

Figure 1. Development of oil palm plantations in Indonesia, Malaysia and Papua New Guinea between 1990 and 2010 (top) and variation in the rate of growth in the different sub-national regions over three temporal periods (bottom).

The trajectory of land use change is fundamentally different in each of the three countries. In Papua New Guinea between 2001 and 2010, only 3% of total deforestation (800,000 ha) was the result of oil palm plantations; nonetheless about 54% of all new oil palm plantations (42,600 ha) originated due to deforestation (see Gunarso *et al.,* 2013 – this publication). In Indonesia, the land use change trajectory is more complex and the forest degradation process is often compounded by wildfire, particularly in Kalimantan, which has led to the development of large areas of quasi-natural habitat dominated by shrubs and grasses (Figure 2). Oil palm plantations have expanded into these so-called "degraded lands" in approximately equal proportions as compared to forest when considering both upland and swamp forest habitats.

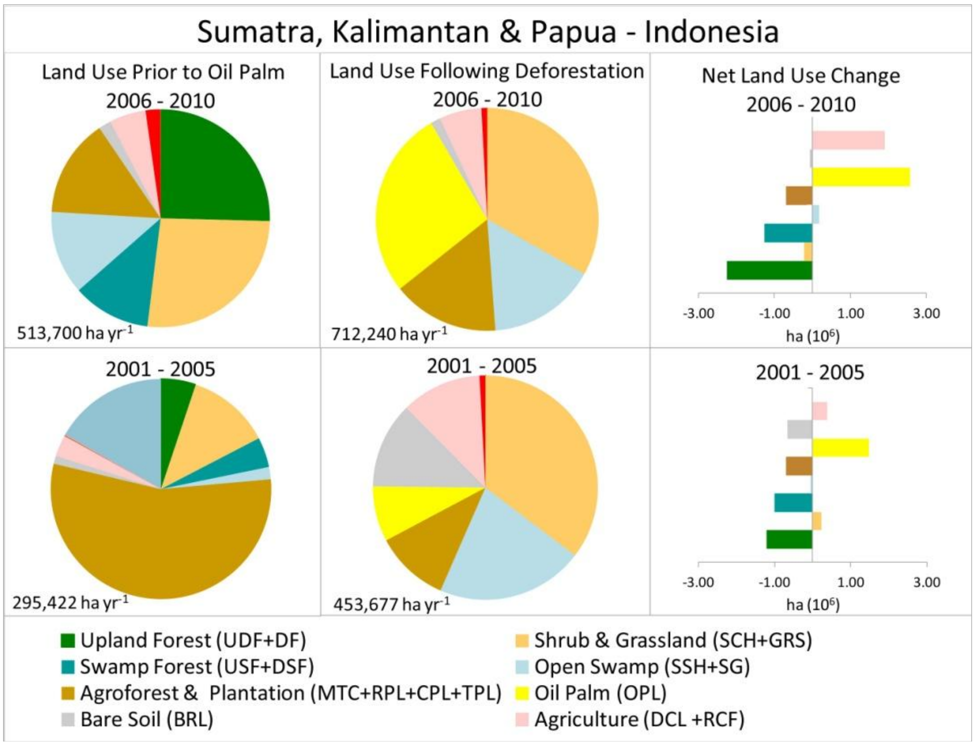

Figure 2. Summary of land use change in the Indonesian territories of Sumatra, Kalimantan and Papua: Left column: land use prior to the establishment of new oil palm plantations (in the lower left corner is the total annual increase in oil palm plantations). Middle column: the fate of land following forest conversion (in the lower left corner is the annual rate of deforestation). Right column: net land use change over each five year period.

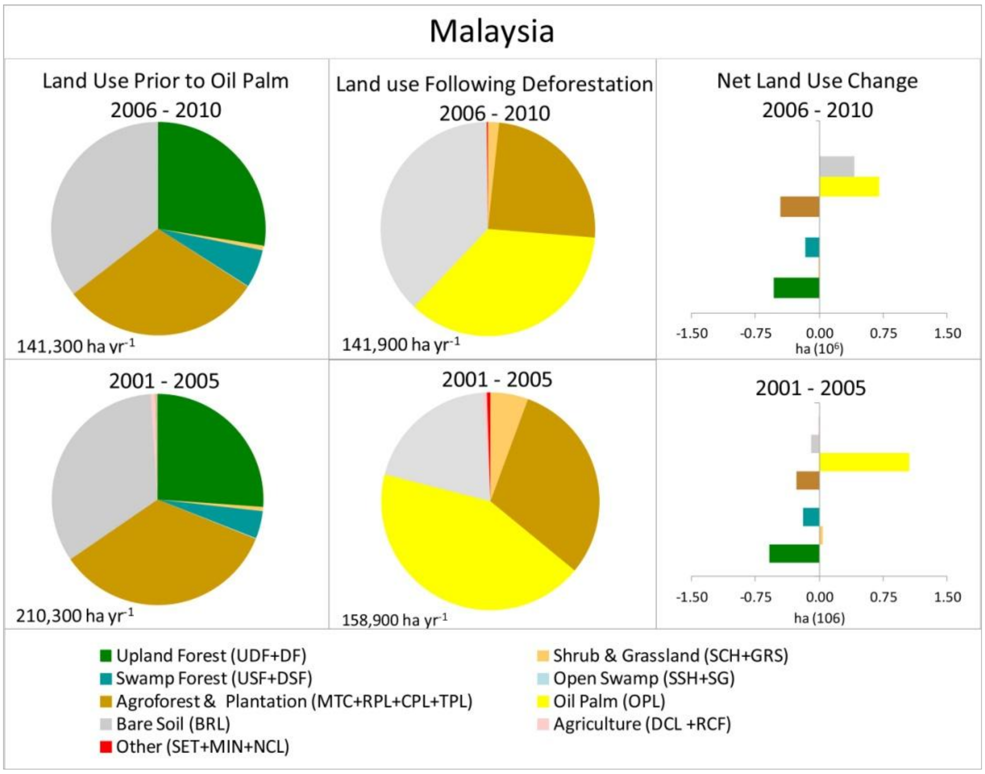

Figure 3. Summary of land use change in the Malaysian territories of Peninsular Malaysia, Sarawak and Sabah. Left column: land use prior to the establishment of new oil palm plantations (in the lower left corner is the total annual increase in oil palm plantations). Middle column: the fate of land following forest conversion (in the lower left corner is the annual rate of deforestation). Right column: net land use change over each five year period.

In Malaysia, the establishment of new plantations tends to be a more straightforward process: Forests are first degraded by intensive logging and although there may be a time lag between logging and conversion, these disturbed forests are then converted directly into oil palm plantations (Figure 3).

In Indonesia and Malaysia, large areas of existing agricultural land and other types of plantation estates were converted to oil palm between 1990 and 2010; at the same time, the agricultural frontier continued to expand at the expense of natural forest landscapes. The total area of the other types of plantations and agroforest decreased, however, because more of these two land cover types were converted to oil palm than were replaced by the conversion of forest (see Figures 2 and 3). The area dedicated to annual crops remained constant in Malaysia, while increasing by about 46% (3.6 Mha) in Indonesia (see Table 6 - Gunarso *et al.,* 2013 – this publication). Land use and land use change is best described as dynamic and complex. Different types of agriculture and plantation production systems are responsible for the conversion of natural forest. A large but variable fraction of deforestation is due to the establishment of new oil palm plantations, which is displacing simultaneously other forms of productive land use. In almost all cases, all forms of agriculture and plantation forestry follow forest degradation, which presumably is initiated by logging and aggravated by wildfire.

The relative proportion of land allocated to oil palm varies among regions. In Malaysia, there is a clear preference to establish oil palm plantations rather than other forms of agriculture and plantation forestry; at the national level, approximately 47% of all productive land (11 Mha) is dedicated to oil palm, a preference that is even more marked in Sabah where 67% of all previously deforested lands (2.3 Mha) are occupied by oil palm (See Figure 4). This trend is reflected also in the land cover category identified as bare soil. Although it is not possible to identify precisely the source and eventual end-use of this land cover type, trends identified in the land use change matrix indicate the preference for oil palm. For example, in Peninsular Malaysia about 9% of bare soil originated from forest landscapes between 2006 and 2009/2010, while 49 % originated from agroforest and other types of plantation landscapes; simultaneously, 6% of bare soils in 2005 were part of the oil palm estate in 2009/2010, a number in line with a replanting cycle of 25 years. In contrast, the previous land cover for bare soil in Sarawak was largely forest habitat, including upland (27%) and swamp habitats (48%). Approximately 50% of all bare soils were eventually planted to oil palm in Malaysia; consequently, the emission estimates for Peninsular Malaysia, Sabah and Sarawak have been adjusted accordingly (see Supplementary Material). In Sumatra, Papua and Papua New Guinea, the category bare soil was employed to identify non-productive land cover types, such as beaches, rock slopes and similar areas, while the bare soils category was not used when classifying land cover in Kalimantan.

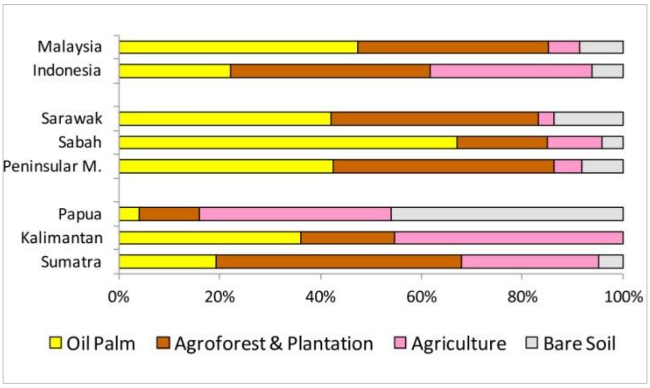

Figure 4. Allocation in 2010 of land dedicated to oil palm, agroforest and other plantations, agriculture and bare soils for Indonesia and Malaysia and the major sub-national regions included in this study. Bare soil is a mixture of exposed substrates, some of which are destined to be converted to one of the other land cover types.

In the three regions of Indonesia included in this study, approximately 24% of productive land cover types dedicated to some type of intensive agriculture, agroforest or plantations estate (32 Mha) have been allocated to oil palm plantations, due mainly to the more diverse productive landscapes that characterize the island of Sumatra (Figure 4). An additional distinguishing characteristics of land cover in Indonesia when compared to Malaysia, is the abundance of quasi-natural non forest habitat categorized as shrub and grassland. These land cover types are often referred to as "degraded lands (Fairhurst & McLaughlin, 2009; Fairhurst *et al.,* 2010) and in 2009/2010 covered an estimated 20% (10.5 Mha) of the total surface area of Kalimantan compared to 5.4% (2.9 Mha) occupied by large scale oil palm plantations (Gunarso *et al.,* 2013 – this publication).

The conversion and drainage of peat soils for the production of palm oil also varies across the region. Sumatra has the largest area of peat soils and the largest area that has been converted to oil palm production (Figure 5).

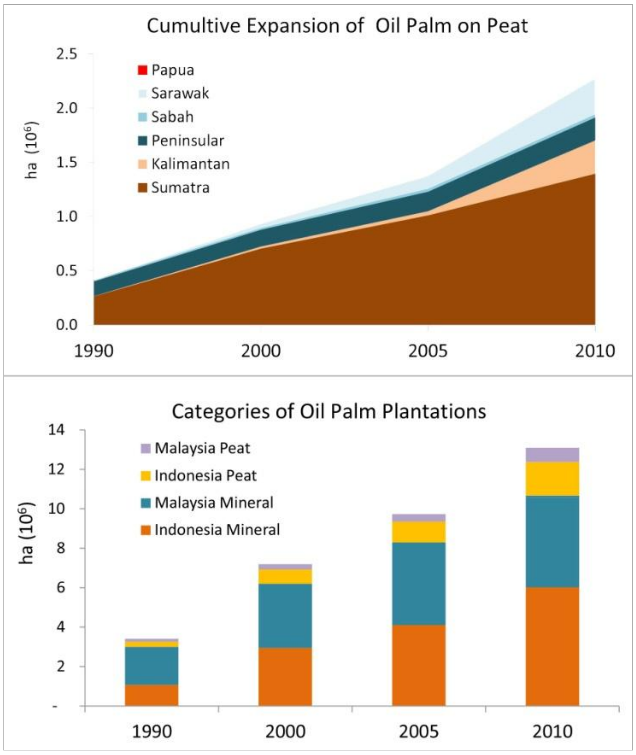

Figure 5. Development of oil palm plantations on peat soils in Indonesia and Malaysia between 1990 and 2010 (top) and the proportion of oil palm on both peat and mineral soils in Indonesia and Malaysia (bottom).

There were about 1.4 Mha or 29% of the total oil palm plantation area in 2009/2010, representing approximately 19% of all the peat soils in Sumatra. Although the overall rate of growth of oil palm in Sumatra decreased in the last temporal period, the rate of conversion of peat swamps increased with an annual rate of conversion that grew from 44,000 in the 1990s to almost 77,000 ha yr^{-1} between 2006 and 2009/2010 (Gunarso *et al.,* 2013 – this publication). Sarawak has the largest proportion (41%) of its total peat swamp area converted to plantations with about 476,000 ha, which also happens to be about 36% of the total oil palm plantation area in the state. Plantations on peat expanded at 59,520 ha yr^{-1} in the last temporal period, translating into an annual loss of 7% of the remaining peat forest habitat in Sarawak (see Supplementary Material, Gunarso *et al.,* 2013 – this publication). Kalimantan converted relatively small areas of peat soil prior to 2005, but converted more than 307,000 ha in the last temporal period, a 10-fold increase in area that represented 11% of all the oil palm plantations on the Indonesian sector of Borneo Island in 2009/2010. Only about 2% of all oil palm plantations in Papua occur on peat, although between 6 and 8 Mha of peat soils have been reported for the region (Wahyunto *et al.,* 2011). Only small areas of peat soils have been reported for Papua New Guinea and there are no reports of oil palm plantations occurring on any of them.

CO_2 Emissions

Net annual emissions from land use change and emissions from peat soils linked to the expansion of oil palm plantations in the study area were estimated at approximately 92 Tg CO_2 yr^{-1} in the first temporal period, which increased to 106 Tg CO_2 yr^{-1} in the second, and then increased markedly to 184 Tg CO_2 yr^{-1} in the most recent period (Figure 6).

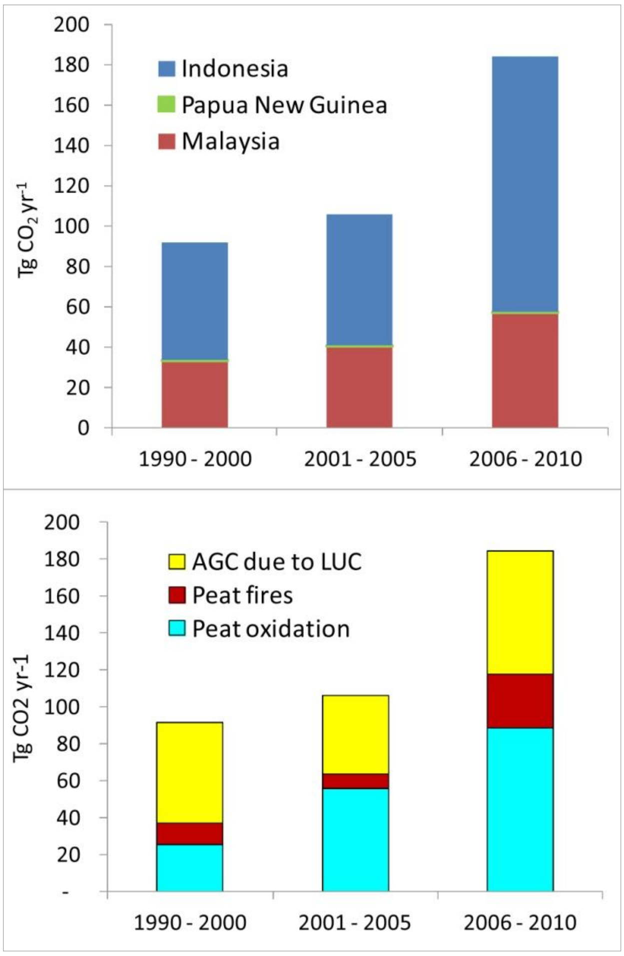

Figure 6. Mean annual emissions stratified by country between temporal periods (top) and (bottom) the same information stratified by source (AGC is above ground carbon and LUC is land use change). Information for Indonesia includes only that for Sumatra, Kalimantan and Papua.

In the three regions of Indonesia included in the study, total net annual emissions from land use in the oil palm sector for the same periods ranged from 58 Tg CO_2 yr^{-1} in the first period, 65 Tg CO_2 yr^{-1} in the second and 127 Tg CO_2 yr^{-1}in the last period. In Malaysia, total net annual emissions for oil palm and land use for the same periods ranged from 33 Tg CO_2 yr^{-1} in the first period, 40 Tg CO_2 yr^{-1} in the second and 57 Tg CO_2 yr^{-1} in the last period. Emissions from Papua New Guinea were estimated at 0.5 Tg CO_2 yr^{-1} between 1990 and 2000, which increased to 0.6 Tg CO_2 yr^{-1} between 2000 and 2010.

The relative importance of the emission source varied over the twenty year period (Figure 6). Between 1990 and 2000 emissions from above ground carbon due to land use change (AGC due to LUC) represented about 60% of total emissions, but emissions from peat oxidation represented 53% of total emissions by the last temporal period. Deforestation as a source of land for the expansion of oil palm became more important in the last temporal period; nonetheless, the incremental emissions originating from existing plantations operating on peat had come to dominate the emission profile. Emissions from peat fires varied over the three temporal periods, essentially tracking land use change on peat soils.

As expected, the total emission profile varied among regions and over time. Sabah, Papua and Papua New Guinea were all characterized by emission profiles dominated by above ground carbon due to land use change, although Sabah's were large when compared to those of Papua and Papua New Guinea (Figure 7). In contrast, the largest source of emissions in Peninsular Malaysia and Sumatra were due to the oxidation of peat, the consequence of declining rates of land use change, but also due to the incremental expansion of oil palm plantations operating on peat soil. Low rates of land use change have stabilized the emissions profile in Peninsular Malaysia, but in Sumatra the relatively large incidence of peat fires indicates that emissions from peat oxidation will continue to increase in the near future. Sarawak and Kalimantan both had emissions profiles that changed over time: AGC due to LUC was the major source of CO_2 emissions in the first period, but the importance of peat oxidation increased as plantations expanded on that soil type. As in Sumatra, the large component of estimated emissions from peat fires is an indication that emissions from peat oxidation will increase over the near term in both Kalimantan and Sarawak (Figure 7).

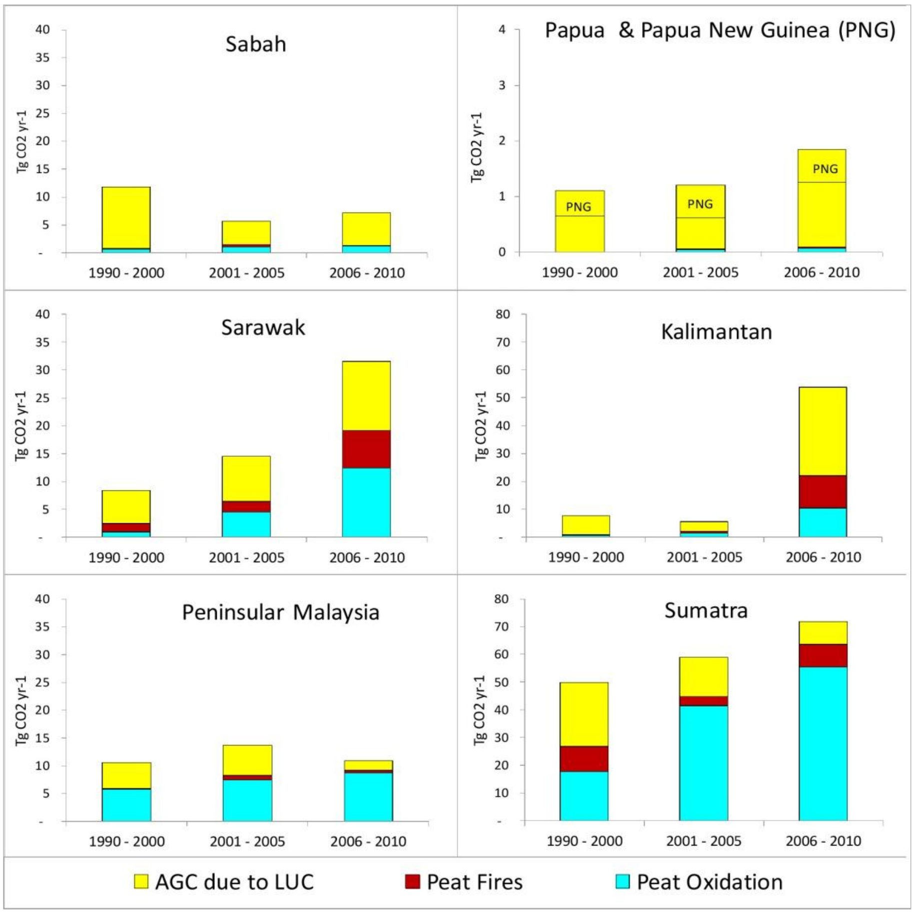

Figure 7. Mean annual emissions stratified by sub region, temporal period and source (AGC is above ground carbon and LUC is land use change); information for Indonesia includes only that for Sumatra, Kalimantan and Papua.

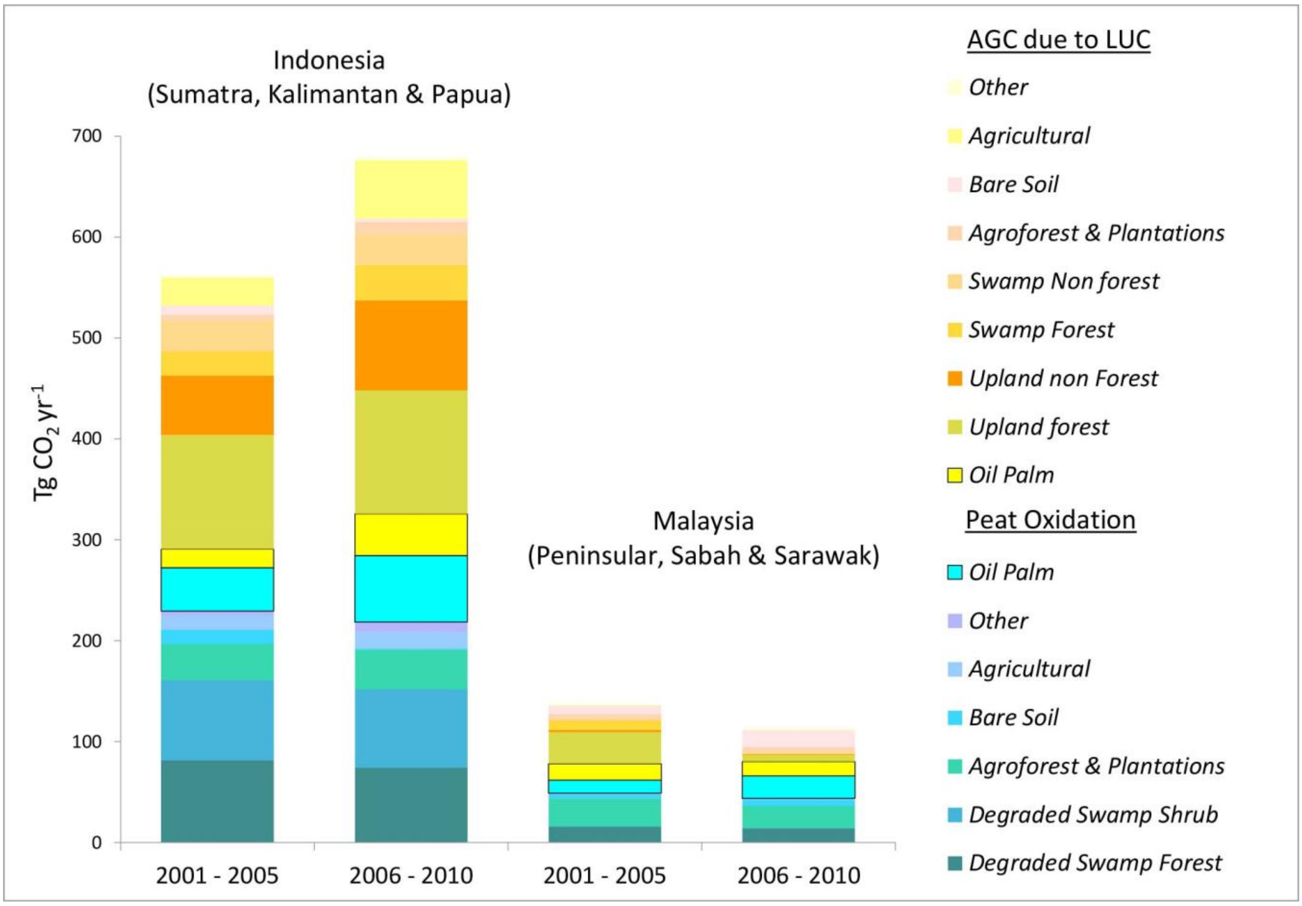

Figure 8. Total mean annual emissions stratified by source of emissions for above ground carbon (AGC) due to land use change (LUC) and the oxidation of peat soils due to drainage and conversion; excludes emissions from peat fires due the lack of fire data for all land cover types.

To evaluate the relative importance of oil palm as a source of CO_2 emissions in the land use sector, we compared emission estimates for oil palm plantations to similar emission estimates for other major land use categories in Malaysia and in the Indonesian study area (Figure 8). This comparison was restricted to emissions from AGC due to LUC and peat oxidation; the impact of peat fires was excluded because of lack of data and a logical framework for developing a model to estimate those emissions. Similarly, only the second and third temporal periods are considered, because we lacked data on land cover change for the other sectors between 1990 and 2000.

Over all, emissions in Indonesia increased from 562 Tg CO_2 yr^{-} in the second temporal period to 679 Tg CO_2 yr^{-1} in the third, with oil palm plantations representing approximately 11% (61Tg CO_2 yr^{-1}) and 16% (107 Tg CO_2 yr^{-1}) of the total from AGC due to LUC and peat oxidation. The largest source of CO_2 emissions came from AGC due to forest degradation with 226 Tg CO_2 yr^{-1} (40%) between 2000 and 2005 and 277 Tg CO_2 yr^{-1} (41%) between 2006 and 2009/2010. The second largest source was peat oxidation from disturbed swamp forests and shrub land, which typically have lower water tables than undisturbed swamp forests due to the construction of canals built to extract timber; our model showed these emissions decreased between the second and third temporal periods from161 Tg CO_2 yr^{-}1 (29%) to 152 Tg CO_2 yr^{-1}(22%), a decline that can be attributed to the conversion of these areas to oil palm plantations, a type of land use change that essentially transfers pre-existing emissions to the palm oil sector (Figure 2). Mean annual emissions from agroforestry and other types of plantations represented about 8% in both periods (43 and 53 Tg CO_2 yr^{-1}), while those from intensive agriculture increased from 7% (42 Tg CO_2 yr^{-1}) to 11% (74 Tg CO_2 yr^{-1}).

Over all, emissions in Malaysia decreased from 136 Tg CO_2 yr^{-1} in the second temporal period to 112 Tg CO_2 yr^{-1} in the third, with oil palm plantations representing approximately 21% (29 Tg CO_2 yr^{-1}) and 32% (36 Tg CO_2 yr^{-1}) of the total amount from both AGC due to LUC and peat oxidation combined. Changes in AGC due to forest degradation were the source of 32% (43 Tg CO_2 yr^{-1}) of total emissions between 2000 and 2005, but decreased to about 8% (8.5 Tg CO_2 yr^{-1}) between 2006 and 2009/2010. Emissions from peat oxidation on degraded swamp forest and shrub habitats decreased from 16 Tg CO_2 yr^{-1} (12%) to 14 Tg CO_2 yr^{-1} (13%); the consequence of these land cover types being converted to oil palm plantations. Annual emissions from agroforestry and other types of plantations declined from 34 Tg CO_2 yr^{-1} (25%) to 27 Tg CO_2 yr^{-1} (24%). The emissions from the AGC due to LUC and peat oxidation linked to intensive agriculture decreased from 0.8 to 05 Tg CO_2 yr^{-1}.

The impact from peat oxidation on the emission profile of oil palm production is becoming increasingly important. Unlike emissions from peat fires and AGC both of which track land use change (Figure 7), the increase in emissions from peat oxidation has been consistent, linear and unidirectional (Figure 6 and 7). The impact of peat oxidation is particularly evident in the case of Sarawak, where it represented less than 11% (0.9 Tg CO_2 yr^{-1}) of total palm oil emissions in the first temporal period, but represented 40% (12.5 Tg CO_2 yr^{-1}) by 2009/2010. Moreover, these statistics do not include the future emissions from an additional 98,000 ha of bare soils on peat documented in the last temporal period (Gunarso *et al.,* 2013 – this publication); historical patterns predict that approximately 80% will be planted to oil palm plantations. Once these lands are incorporated into the oil palm estate, our models predict that emissions from peat oxidation will increase by approximately 30% in Sarawak. Sumatra has an even greater legacy of long-term CO_2 emissions from peat oxidation, which represented 77% (56 Tg CO_2 yr^{-1}) of total emissions linked to oil palm plantations for the island by 2009/10.

In Peninsular Malaysia, where approximately 8% of oil palm are operating on peat soils (Gunarso *et al*. 2013 – this publication) they are now the source of about 84% (9 Tg CO_2 yr^{-1}) of the emissions profile of oil palm linked to land use, a statistic that is not likely to change significantly over the short term. In the case of Kalimantan, AGC due to LUC remains the predominant source of emissions, but emission of peat oxidation will increase in the short term. Only Sabah shows consistently low levels of emissions from peat oxidation, due to the relative scarcity of peat soils in that state. Over all seven regions, plantations operating on peat soils occupied about 18% (2.4 Mha) of the spatial footprint of large-scale oil palm plantations, but peat oxidation from these plantations represented 48% (88 Tg CO_2 yr^{-1}) of the total emission profile in 2009/10.

DISCUSSION

This report provides the first sector-wide estimate of CO_2 emissions linked to land use and land use change for the palm oil industry in the geographic region that produces 85% or more of the world supply of palm oil and palm oil products (Teoh, 2010). The primary objective of this report was to estimate the sources, dimensions, and trends of emissions over the past twenty years; as a secondary objective, we compared these emissions in the broader context of emissions caused by other types of land use. Previous reports of CO_2 emissions linked to land use and palm oil have either been based on bottom up models that estimate emissions as a function of palm oil mass or unit of energy (Reijnders & Huijbregts, 2008; Wicke *et al.,* 2008) or on landscape-scale analyses that do not provide a global estimate of emissions, nor capture the geographic variability characteristic of the industry (Uryu *et al.,* 2008; Koh *et al.,* 2011; Carlson *et al.,* 2012a; 2012b; Miettinen *et al.,* 2012a, 2012b). This report provides detailed information on the historical emissions linked to the expansion and operations of oil palm plantations stratified according to land cover source, soil type, geographic region, and temporal period. This information is essential for establishing the industry's baseline emissions and for developing future scenarios to evaluate the impact of different development options (see Harris *et al.* 2013 – this publication).

Land Cover Classification

Like all studies that document the complex phenomenon of land use and land use change, our study addressed the challenges linked to the quality of available data and the difficulties of interpreting dynamic processes that change over time. These challenges, and the decisions on how to manage them, are sources of variation and uncertainty inherent in a study of this nature. For example, the stratification of land cover types into undisturbed forest, disturbed forest, shrub land and grassland is an approach used by ecologists to qualitatively describe a continuous gradient; however, deciding where one category ends and the next begins is imprecise, and sometimes arbitrary, particularly when relying on satellite imagery covering large heterogeneous areas characterized by varying levels of human activity. Many academic studies choose to manage this challenge by using automatic classification techniques based on the spectral signature of image pixels (Hansen *et al.,* 2009; SarVision, 2011; Broich *et al.,* 2011; Carlson *et al.,* 2012b; Miettinen *et al.,* 2012a; Margono *et al.* 2012), but that approach limits the number of categories that can be discriminated and excludes useful information that can be reasonably interpreted from the landscape context. Moreover, as the number of strata increase, automatic procedures require extensive human editing, which in terms of labour and objectivity, are not unlike visual recognition techniques. Fortunately, oil palm plantations are easy to identify in satellite imagery and the results from Gunarso *et al.* (2013 – this publication) are similar to other studies that have been conducted on shared landscapes (see below). The same cannot be said for the ability to distinguish among other natural, quasi-natural and human-derived land cover types, however, and the level of confidence in the transitions among these different land cover types is less robust. To improve accuracy and facilitate communication, we aggregate similar types of land cover categories based on edaphic attributes (upland vs. swamp), vegetation type (forest vs. shrub and grassland), and land use (plantation and agroforest vs. agriculture).

Land Use Change and Above Ground Carbon

One of the largest sources of uncertainty in estimating the emissions from land use change linked to oil palm plantations is the variability in carbon stock estimates in above ground carbon for the different land cover types. The source of this variability has three origins: 1) natural spatial variability of AGC in forest and non forest land cover types, 2) the impact of logging and fire on above ground carbon in intact but disturbed forest habitats, and 3) the temporal period which is used to calculate emissions from land use change.

For forest, shrub and wetland categories, we use the mean value of all published reports from Indonesia, Malaysia and Papua New Guinea, while values for agriculture, agroforest and other plantation categories were based on scientific and technical publications (see Agus *et al.,* 2013 – this publication). Agroforest, which is sometimes referred to as mixed tree crops in the Indonesian classification system, is a heterogeneous category of different land use intensities, including secondary forests, small farms, pastures, coffee and cocoa, and even small-scale oil palm plantations. The border between agroforest, disturbed forest and shrub land is subject to interpretation and, consequently, a source of uncertainty in emissions estimates.

The estimates of the carbon stock in oil palm plantations, which represent a uniform cropping system and a species with simple allometry, have also been the subject of discussion among workers who seek to estimate the GHG footprint of palm oil as part of a life cycle analyses. For example, a fully mature 25-yr old plantation can have as much as 155 Mg C ha^{-1}, while time-averaged estimates range from 23 to 50 Mg C ha^{-1} (Dewi *et al*., 2009; Khasanah *et al.,* 2011). The time-averaged value adopted in this study (36 Mg C ha^{-1}) does not account for differences among new high yielding dwarf varieties or short rotation cycles favored by some companies, nor low stand densities in poorly managed plantations, senile plantations on peat soils, or smallholder's crops that might have a low carbon stock value.

Soil type and climate influence plant growth and lead to differences in AGC in humid, semi-humid and dry forest formations (Saatchi *et al.,* 2011). Carbon stocks are also influenced by species composition and the *Dipterocarpaceae*, a plant family that dominates many forests in Southeast Asia, is characterized by tall trees with high wood density which endows undisturbed forests in the study area with unusually high values for above ground carbon (Slik *et al.,* 2009). The relative abundance of this family, which is also known for its high quality timber, also influences logging intensity; and timber extraction rates in Borneo have been estimated at 230 m^3 ha^{-1} — an order of magnitude greater than is common to Amazonian forests (Butler, 2009). This level of logging intensity reduces the carbon stocks in a standing forest, and is a major cause of forest degradation that is magnified by conventional logging practices (Sist *et al.,* 2003). In spite of the loss of above ground carbon, the logged forests in Southeast Asia retain much of their original biodiversity and as many as 75% of the original complement of birds and dung beetles persist in disturbed forests (Edwards *et al.,* 2010). The innate value of this biodiversity, coupled with the inherent capacity of these forests to regenerate and restore carbon stocks, motivate some ecologists and environmental advocates to refer to these disturbed forest as "natural forests" or "intact tropical forests" or "primarily intact forests" or even the oxymoronic "degraded primary forests."

Some ecologists and many foresters use the term "secondary forest" to describe disturbed and degraded forests; this term has its origin in classic ecological theory that describe how ecological processes mediate a succession of vegetation types following severe disturbance (Clements, 1916). The terms "secondary forest" and "degraded forest" are used by advocates of the palm oil sector to emphasize that palm oil expansion has not occurring at the expense of "primary forests," an affirmation supported by the land use change study that underpins this report (Gunarso *et al.,* 2013 – this publication). This view emphasizes the economic advantages of palm oil production in the context of the low residual economic value of intensively logged forests, the contribution of palm oil to national GDP and its benefits to rural livelihoods (Cramb & Cury, 2012).

We avoid these pitfalls in terminology by using the terms "disturbed" and "undisturbed" forest, as well as document the transition from undisturbed forest to disturbed forest, and then to shrub and grassland, with separate categories for both upland and wetland habitats (Table 1). In addition, we relied on five year temporal comparisons to capture the intermediate stages that distinguishes our study from others that used longer temporal periods (Koh & Wilcove 2008; Carlson *et al.,* 2102b; see Discussion in Gunarso *et al.,* 2013 – this publication). Unfortunately, we were not able to fully document the changes in land cover change between 1990 and 2000 in Indonesia when both logging and forest conversion were at their highest (Hansen *et al.,* 2009); nonetheless, evidence from the two subsequent periods shows that the oil palm sector is not responsible for the loss of the largest part of the carbon stocks of the original forest cover in these regions (Figure 9). Forest loss via degradation was greatest in Kalimantan where 40% of forest loss between 2006 and 2009/2010 was caused by the degradation of approximately 0.9 Mha of forest to shrub land and the release of 155 Tg CO_2 yr^{-1}, almost 52% of total emissions for the region excluding all emissions from peat fires. The historical emissions from above ground carbon due to forest degradation, presumably due to logging and wildfire, were more than four times greater than emissions from above ground carbon due to land use change caused by the establishment of new oil palm plantations in the same temporal period (32 Tg CO_2 yr^{-1}).

Emissions from Peat

Emissions from peat oxidation and peat fires have increased in both absolute and relative terms over the 20 year period and now represent a total of 64% (118 Tg CO_2 yr^{-1}) of all emissions from land use and land use change linked to the palm oil sector. If the one-time emissions from peat fires are excluded, then emissions from peat oxidation represent 48% (88 Tg CO_2 yr^{-1}) of total emissions. Moreover, CO_2 emissions from peat oxidation are not subject to the temporal fluctuations linked to land use change and the establishment of new plantations. Unless these plantations are abandoned and restored as wetlands, they represent a long-term attribute of the palm oil production system (Schrier-Uijl *et al.,* 2013 – this publication). Although the direction and trend of CO_2 emissions from peat oxidation are clear, the actual dimensions of these emissions remain uncertain. This uncertainty is the consequence of four factors: 1) the spatial extent of peat soils, 2) the depth of drainage, 3) the rate of oxidation of peat, and 4) the incidence of fire at the time of plantation establishment (see Agus *et al.,* 2013; Schrier-Uijl *et al.,* 2013 – this publication).

The spatial data used to model emissions in this and other studies are based on soil maps derived from satellite imagery, and thus are subject to the uncertainty linked to that technology. Gunarso *et al.* (2013 – this publication) had access to two sources of information on the distribution of forest wetland: a peat soil map distributed by Wetlands International for Indonesia (Wahyunto & Subagjo, 2003; Wahyunto & Suparto, 2004; Wahyunto *et al.,* 2006) and data from the Harmonized World Soil Database for Malaysia (FAO 2009). However, a more recent study for Sumatra and Kalimantan has reduced the spatial extent of peat swamps by approximately 15% (Wahyunto *et al.,* 2011), while a study using official soil maps developed for Malaysia reported that peat formations were 5% greater (Omar *et al.,* 2011). Since the emissions from peat are dependent on a model that uses data derived from these information sources, improvements in the accuracy and precision will impact estimation of emissions from peat fires and peat oxidation.

Assumptions made regarding the depth of drainage impacts the outputs from models that estimate CO_2 emissions due to peat oxidation. According to better management practices recommended by the Roundtable for Sustainable Palm Oil, the recommended depth of drainage is 60 cm, a level which both maximizes plant productivity and minimizes CO_2 emissions. In many plantations, water table depths are not actively managed and often fall below 80 cm during the annual dry season, particularly during periods of severe drought (Lim *et al.,* 2012). Since the models used to estimate emissions from peat oxidation are simple linear correlations, the mean level of drainage used in those equations will directly impact emissions estimates.

The heterotrophic respiration linked to the degradation of the peat, here referred to as peat oxidation, is perhaps the most uncertain of all the emission factors used to model emission estimates from oil palm plantations. Studies conducted over the past decade have generated estimates of heterotrophic respiration that range from 20 to 95 Mg CO_2 ha^{-1} yr^{-1} (see review in Agus *et al.,* 2013 – this publication). The differences stem from methodological challenges associated with the two main experimental approaches employed to measure peat oxidation. One approach correlates soil subsidence with peat oxidation, a method that can confound soil compaction with peat degradation and, consequently, requires research protocols that document bulk density (weight per volume) and carbon density (% carbon content). The other approach directly measures CO_2 flux on the soil surface using closed chamber systems; however, this method must discount for autotrophic respiration from plant roots, which produce CO_2 while consuming carbohydrates produced by photosynthesis in the leaves of living plants. Failure to adequately account for autotrophic respiration will inflate estimates of CO_2 emissions from peat oxidation. The selection of 43 Mg CO_2 ha^{-1} yr^{-1} was based on a review of recent studies and is near the median value of the range of these values (see Agus *et al.,* 2013 – this publication); other recent studies have based their models on a substantially higher emission factor of approximately 95 CO_2 ha^{-1} yr^{-1} (Uryu *et al.,* 2008; Koh *et al.,* 2011; Carlson *et al.,* 2012a; 2012b; Miettinen *et al.,* 2012a, 2012b)

Peat fires are an important source of CO_2 emissions in Southeast Asia and the haze linked to those fires is an important transboundary issue within the region. Estimation of historical emissions from peat fires has high uncertainty, because of the difficulty in documenting the intensity, depth and spatial extent of fire data collected by satellite sensors. For example, modeled estimates of CO_2 emissions during the unusually severe *El Niño* event of 1997/98 produced values between 2.9 and 9.4 Pg CO_2 when extrapolated across all of Indonesia (Page *et al.,* 2002). A similar approach that included fires in both mineral and peat soils reported emissions of 3.5 Pg CO_2 for the same event, as well as estimating annual emissions from fire in Southeast Asia that fluctuated between 0.09 and 1.3 Pg CO_2 between 2000 and 2009 (van der Werf *et al.,* 2010). We did not calculate region-wide estimates of peat fire emissions due to lack of data on the distribution and severity of peat fires across all land cover types. Our modeled estimates of historical emissions from peat fires for oil palm plantations are based on the assumption that differential amounts of peat are consumed by fire at the time of plantation establishment from forest and shrub (see Agus *et al.,* 2013 – this publication). Our estimates of emissions from peat fires on oil palm plantation correspond to 2% of total mean annual fire emissions between 2000 and 2005 (481 Tg CO_2 yr^{-1}) and 6% between 2005 - 2009/2010 (467 Tg CO_2 yr^{-1}) (van der Werf *et al.,* 2011 and Supplementary Material).

The Impact of Uncertainties

Taken individually, the variability of any single emission factor can lead to relatively large differences in the final estimate of the CO_2 emissions; taken together, these uncertainties become multiplicative and lead to very different estimates of the carbon footprint of palm oil (Reijnders & Huijbregts, 2008). Based on published reports, the range of potential carbon stock values in forest land cover types is from 74 to 360 Mg C ha^{-1}, the emission from peat oxidation may be half as much smaller or twice as large, and the potential depth of burning can vary from zero to as much as 50 cm depending on the severity of seasonal drought. The values selected for the modelled estimates presented here are based on the mean value of all published peer reviewed studies (above ground carbon), a critical evaluation of peer reviewed studies (peat oxidation) and recommendations from informed individuals (peat fire depth).

A comparison of a subset of our results based on land use change data from Kalimantan (Gunarso *et al.* 2013 – this publication) with a similar study focusing on palm oil and CO_2 emissions from the same region (Carlson *et al.,* 2012b) provides an opportunity to evaluate how different emission factors, land cover stratification methodologies, and temporal perspectives impact model outputs. Both studies were based on land use change data derived from similar satellite imagery covering two decades between 1990 to 2009/10. Both are in close agreement as to the rate of growth of oil palm plantations (293% for Gunarso *et al.* vs. 278%, for Carlson *et al.*). Both have similar estimates of the spatial footprint of oil palm plantations in 2009/10 (2.9 vs. 3.2 Mha), and both arrive at similar estimates of the total area of oil palm plantations established on peat soils in 2009/10 (307,000 vs. 402,000 ha). However, the two studies have very different emission estimates (Table 3). Understanding the source of these differences is essential for organizing emission monitoring protocols that will allow the palm oil sector to accurately quantify its CO_2 emissions, as well as identifying strategies to reduce those emissions.

Table 3. Emissions from land use and land cover change from oil palm plantations in Kalimantan

	Carlson *et al.* 2012b Mg CO_2	Agus *et al.* 2013 Mg CO_2
1990 - 2000		
AGB from LUC	309,138,862	65,802,767
Peat oxidation	18,219,572	6,062,943
Peat fires	17,360,229	4,230,649
Total	**344,718,663**	**76,096,360**
2000 - 2010		
AGB from LUC	906,122,095	176,767,485
Peat oxidation	250,194,189	59,466,820
Peat fires	257,480,905	61,354,254
Total	**1,413,797,189**	**297,588,558**

In the first temporal period, the expansion of oil palm plantations on peat soil was relatively small; consequently, the difference in the emissions estimates is due largely to assumptions regarding how land cover classes were defined and how land use change was quantified. Carlson *et al.* (2012b) recognized two forest classes, agroforest and non forest, while Gunarso *et al.* (2013 – this publication) recognized four forest types and four non forest types, as well as separate agroforest and plantation categories. The relative abundance of these categories and their associated carbon stock values was the source of 91% of the variance in the emissions profiles between the two studies (see discussion in Gunarso *et al.,* 2013 – this publication). There is an element of subjectivity to any land cover classification, particularly when attempting to stratify a continuous gradient, which in this case is a transition from undisturbed forest to grassland. In that context, Carlson *et al.,* (2012b) recognized more area as forest along that gradient, while Gunarso *et al.* (2013 – this publication) recognized more area as shrub and grassland.

An alternative methodology is to use pixel-based estimates of carbon density that reflect the variability of ecological gradients (Saatchi *et al.,* 2011). In a companion study, Harris *et al.* (2013 – this publication) used this type of information to model future emissions scenarios for different oil palm development strategies. As part of that effort, they used the polygons developed by Gunarso *et al.* (2013 – this publication) in combination with the pixel-based data from Saatchi *et al.* (2011); their objective was to train the forward looking model using historical land use change data between 2000 and 2010 (see Table 4 in Harris *et al.,* 2013- this publication). That training exercise revealed that the AGC stock values selected for the four forest habitats were similar to the mean values derived from the pixel-based map of carbon density (see Table 4, Harris *et al.,* 2013 – this publication). In contrast, the mean values selected for AGC for shrub categories were about 50% lower for upland habitats and 25% lower for swamp habitats. If we had used mean carbon stock values for shrub land similar to those derived from pixel-based values, the modelled emission estimate from above ground carbon due to land use change in Kalimantan would have increased by about 35 Tg CO_2 yr^{-1} (a 40% increase) between 2000 and 2005 and 86 Tg CO_2 y^{-1}(a 50% increase) between 2006 and 2009/10. Nonetheless, these modified values would still be less than 50% of the modelled estimates reported by Carlson *et al.* (2012b) (see Table 3).

In the temporal period spanning 2001 to 2009/2010, the source of variance is more complex with 65% of the difference attributed to AGC due to LUC with the remaining variance originating from the use of different emissions factors for peat: 17% from peat oxidation and 18% due to peat fire. In the case of peat oxidation, the major factor was the selection of an emission factor of 95 Mg CO_2 ha^{-1} yr^{-1} by Carlson *et al.* (2012b) versus a value of 43 Mg CO_2 ha^{-1} yr^{-1} recommended by Agus *et al.* (2013 – this publication). Similarly, Carlson *et al.* (2012b) assume that on average 203 Mg C ha^{-1} are lost during a fire event on peat soils, while Agus *et al.* (2013 – this publication) recommended values of 90 Mg C ha^{-1} lost from peat soil fires from forest conversion and 30 Mg C $ha^{-1}$from peat soil fires on shrub land. The difference in the modelled estimates are the consequence of the assumption made concerning the depth of peat fires: Carlson *et al.* (2012) assumed a mean burn depth of 33 cm based on studies documenting the impact of fire during *El Niño* drought years (Ballhorn *et al.,* 2009), while *Agus et al.* (2013 – this publication) assumed that on average 15 cm are lost when forest is cleared and burned and 5 cm when shrub land is cleared and burned.

Finally, the time frame in which the comparison is made is an additional factor that can influence the estimation of CO_2 emissions and, subsequently, allocating those emissions to the appropriate economic or social actor. Between 2000 and 2010, Gunarso *et al.* (2013 – this publication) stratified LUC into two five year periods (2000 to 2005 and 2006 to 2009/2010), while Carlson *et al.* (2012b) evaluated change between 2000 and 2010. As the authors point out in the supplementary information of their article: "Due to the 10-year interval between the land cover product and the oil palm coverage, our analysis likely overestimates the amount of intact forest converted to oil palm" (see Supplementary Information, page 7 from Carlson *et al.,* 2012). The adoption of two five year periods allowed Gunarso *et al.* (2013 – this publication) to document the sequential degradation of undisturbed forest to disturbed forest and then to shrub land prior to its conversion to oil palm plantations (see Figure 10, Gunarso *et al.,* 2013 – this publication). The recognition that land is degraded and partially depleted of carbon stocks prior to its conversion to oil palm plantations should be taken into account when estimating the CO_2 emission profile of palm oil. At least some of those historical emissions are more properly allocated to the forest sector due to the intensive logging regimes that characterize the region (Putz *et al.,* 2008) and the impact of forest fires on peat soils that are a combination of bad luck due to drought and the difficulty in fighting wildfires in remote regions of Indonesia (van der Werff *et al*, 2010).

CONCLUSIONS

The rate of expansion of oil palm plantations has been remarkably constant at approximately 7% per annum from 3.5 to 13.1 Mha between 1990 and 2010. The growth in the spatial extent of oil palm plantations has been accompanied by a concomitant increase in the CO_2 emissions, which including all CO_2 emissions from AGC due to LUC, peat oxidation and peat fires, has grown from 92 Tg CO_2 yr^{-1} between 1990 and 2000 to 106 Tg CO_2 yr^{-1} between 2001 and 2005 and 184 Tg CO_2 yr^{-1} between 2006 – 2009/2010. In the third temporal period, 67 Tg CO_2 yr^{-1} (36%) originated from AGC due to LUC and about 90% of these emissions came from deforestation, which has been the source of about 3.5 Mha of the land that has been used for the establishment of new plantations. A smaller area of approximately 3.3 Mha originated on landscapes classified as agroforest or other types of plantations, while 1.7 Mha was developed on land that had been covered by forest in 1990, but which had been degraded to shrub and grassland prior to its conversion to oil palm plantations between 2000 and 2010.

The documentation of this land use trajectory, which includes the transition from undisturbed forest to disturbed forest to shrub land and eventually grassland, dominates the historical CO_2 emissions of the region. Forest degradation, presumably due to intensive logging and its subsequent conversion to shrub land due to wildfire, contributed approximately five times greater emissions (285 Tg CO_2 yr^{-1}) between 2006 and 2009/2010 than the AGC due to LUC component of the palm oil emissions profile (55 Tg CO_2 yr^{-1}) for the same period. This explains, in part, why our estimates of oil palm emissions from AGC due to LUC are less than a third of other studies whose models assume that oil palm plantations are established on forests landscapes of high carbon density. The results from a companion article (e.g., Gunarso *et al.,* 2013 – this publication) show that the land cover types used for oil palm plantation expansion has varied over time and among geographic regions; emissions from AGC due to LUC, not surprisingly, track those differences. Emissions from AGC due to LUC can be reduced by promoting oil palm plantation expansion on landscapes with low to moderate levels of AGC, such as the approximately 9 Mha of shrub and grassland in Kalimantan and 8 Mha of agroforest in Sumatra (see Gunarso *et al.,* 2013 – this publication).

Plantations on peat soils now represent about 18% of the spatial footprint of the palm oil industry (2.4 Mha), but represented almost 64% (118 Tg CO_2 yr^{-1}) of the total CO_2 emissions profile in the last temporal period. About 16% (29 Tg CO_2 yr^{-1}) are linked to peat fires, while almost 48% (88 Tg CO_2 yr^{-1}) originate from peat oxidation from existing oil palm plantations operating on peat soils. The emissions from peat fires are one-time events that occurred in the past when forests and shrub land were cleared for new oil palm plantations; these fires are now illegal and unlikely to contribute to future emission profiles. In contrast, emissions from peat oxidation will continue to grow in absolute terms as oil palm companies develop new plantations on existing concessions on peat soils in Sumatra, Kalimantan and Sarawak. Even if the industry acts to halt new development on peat soils, the existing oil palm plantations on peat soils will continue to emit CO_2 at approximately these levels for the foreseeable future. Emissions from peat oxidation can only be terminated by restoring the natural hydrological and ecological conditions that cause peat to form in the first place. Similarly, enforcing the ban the use of fire for land clearing will significantly reduce emissions, especially on peat land.

Just as CO_2 emissions from AGC due to forest degradation are greater than those linked to land use change from the palm oil sector, emissions from peat oxidation from degraded swamp forest with altered hydrological regimes are greater than similar emissions from oil palm plantations (166 vs. 88 Tg CO_2 yr^{-1}). Emissions from degraded swamp forests have declined in the last temporal epoch, in part because logging of remnant swamp forests already has declined, but also because this land cover type is being converted into oil palm plantations. Consequently, CO_2 emissions from degraded swamp forests are being transferred from that category to the oil palm plantation sector.

Finally, by comparing our results with other recently published studies, we show that the uncertainties in estimating CO_2 emissions are subject to the methodological approaches and assumptions used to model emissions from land use and land use change (see review by Agus *et al.,* 2013 – this publication). In spite of the differences in the dimensions of the CO_2 emissions between our models and those employed by other studies (see Carlson *et al.,* 2012b; Page *et al.,* 2011), the overall trends are nearly identical. The rapid expansion of palm oil sector over the last two decades has been responsible for the emissions of several gigatons (Pg) of CO_2 from land use and land use change. Understanding the sources of these emissions, which have been variable in time and space, is a necessary first step in identifying strategies for reducing, eliminating or even reversing the net CO_2 emissions of the industry.

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PROJECTIONS OF OIL PALM EXPANSION IN INDONESIA, MALAYSIA AND PAPUA NEW GUINEA FROM 2010 TO 2050

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ABSTRACT

Due to the growth of the Southeast Asian palm oil industry over the last decade and growing concern over its potentially negative environmental impacts, quantification of oil palm expansion and associated carbon emissions has become increasingly important. Here we simulate future scenarios of oil palm expansion until the year 2050 across Indonesia (Kalimantan, Sumatra and Papua), Malaysia (Sarawak and Sabah) and Papua New Guinea. We investigate the potential magnitude of net carbon emissions under three policy scenarios: (1) expansion of the industry to double production by the year 2050, which assumes that growth will follow practices defined as business as usual (BAU); (2) a moratorium on peat coupled with yield improvements of 0.7% annually, which reduces the demand for land and limits new oil palm expansion to low biomass landscapes on mineral soils (MRT); and (3) a moratorium on peat coupled with yield improvements – *plus* the gradual displacement of existing plantations on peat to low biomass areas on mineral soils, starting in 2020, with subsequent rewetting and restoration of retired plantations to natural peat forest (RET).

Net cumulative carbon emissions under BAU are estimated to be 15.2 Pg CO_2 by 2050; approximately 77% of these emissions would originate from the continuous drainage of peat on existing and new plantations, which by 2050 would cover 15% of the total area of oil palm plantations estimated at 26 Mha. Halting expansion into peat areas and shifting it to lower biomass areas in the MRT scenario can potentially reduce total net cumulative emissions by more than 50%. Displacing existing plantations on peat to mineral soils, rewetting drained peat and restoring retired peat plantations to native forest vegetation in the RET scenario would eventually lead to annual emissions near zero for a mature stable oil palm sector covering approximately 21 Mha of plantations.

Key words: *peat, deforestation, GEOMOD, carbon, restoration, scenarios*

INTRODUCTION

Southeast Asia is the world's largest producer of palm oil, with Indonesia and Malaysia producing approximately 87% of the global supply (Koh *et al.,* 2011). Production has increased at a rate of approximately 7% per year over the last two decades, with the area of mature oil palm in Indonesia, Malaysia and Papua New Guinea increasing from 3.5 million ha in 1990 to 13.1 million ha in 2010, with 7.7 Mha in Indonesia, 5.2 Mha in Malaysia and 0.134 Mha in Papua New Guinea (Gunarso *et al.,* 2013 - this publication). Palm oil is now the world's most widely used vegetable oil and the area under cultivation is expected to increase further as demand grows for use in food, consumer goods and biodiesel (USDA, 2009; USDA, 2010). Indonesia is expected to increase its production to meet most of this demand. The top three expansion areas highlighted by the Indonesian Ministry of Forestry and Planning in 2004 were Kalimantan (10.3 million ha), Sumatra (7.2 million ha), and Papua (6.3 million ha) (USDA, 2009). Malaysia's palm oil growth is expected to decline due to limited available area, while investment is expected to increase in less developed regions of Southeast Asia, such as Papua New Guinea, where Malaysian companies recently announced planned investments in 150,000 ha of expanded oil palm plantations (Koh & Wilcove, 2009; USDA, 2010).

While oil palm is a pillar of Indonesia's and Malaysia's economies and represents a growth opportunity for countries like Papua New Guinea, this development comes at a substantial environmental cost. Palm oil is a major driver of deforestation in these regions (Carlson *et al.,* 2012), and tropical deforestation has been shown to account for approximately 7-14% of total global anthropogenic carbon emissions (Harris *et al.,* 2012). Indonesia and Malaysia account for 80% of Southeast Asia's remaining primary forest that provides vital ecosystem services, supporting high biodiversity, water quality, disaster management, and carbon sequestration (Fitzherbert *et al.,* 2008), yet Indonesia and Malaysia have experienced the highest loss of forest of all Southeast Asian countries, with Indonesia second only to Brazil as the world's largest emitter of CO_2 from deforestation between 2000 and 2005 (Harris *et al.,* 2012).

Due to the rapid growth of the palm oil industry over the last decade, the need for accurate estimates of its expansion and associated carbon emissions has become increasingly important. This is particularly so when oil palm expands into forest with high carbon stocks or into peat swamp areas that emit large quantities of carbon when the peat is drained for plantation development. Proper carbon flux accounting requires reliable spatial and temporal mapping of oil palm and other dominant land cover types for tracking oil palm expansion pathways (Hansen *et al.,* 2009; Koh *et al.,* 2011; Paoli *et al.,* 2011). Knowledge of these land cover changes is important to provide government, private and civil society with the information needed to improve practices and policies governing land use. Despite growing concern over the last decade of the potential negative environmental impacts of oil palm expansion, accurate mapping of oil palm across large regions of Southeast Asia has not been achieved until recently (Stibig & Malingreau, 2003, Miettinen *et al.,* 2011, Miettinen *et al.,* 2012a, Miettinen *et al.,* 2012b, Broich *et al.,* 2011, Margono *et al.,* 2012; Gunarso *et al*., 2013 - this publication).

In an analysis of oil palm expansion into peat areas across Peninsular Malaysia, Borneo and Sumatra, Koh *et al.* (2011) used moderate resolution satellite imagery to demonstrate that approximately 800,000 of tropical peat land in the study region had been converted to oil palm by 2000, representing 6% of the total peat swamp area in the study area, and that closed canopy oil palm occupied approximately 8.3 Mha, with 47% in Sumatra, 29% in Borneo and 24% in Peninsular Malaysia. The study highlighted concern for the expansion of oil palm on peat soils in the provinces of Riau in Sumatra and in Central and West Kalimantan in Borneo. However, by focusing only on peat land, the study was limited in addressing the full scope of land cover change.

In a later study, Carlson *et al.* (2012) analyzed land cover change pathways and resulting net carbon emissions in Kalimantan from 1990 to 2010 using higher resolution imagery (30 m spatial resolution), and projected a future expansion of 9.4 Mha by 2020, based on the assumption that the entire area within existing concessions would be converted eventually into oil palm plantations. They estimated that this expansion would lead to an 18-22% contribution to Indonesia's overall carbon emissions by 2020.

Here, we build on these analyses as well as that of Gunarso *et al.* (2013 – this publication) and Agus *et al.* (2013b – this publication) to simulate future scenarios of oil palm expansion across the study regions of Kalimantan, Sumatra and Papua in Indonesia, Sarawak and Sabah in Malaysia, and in Papua New Guinea up to the year 2050. We investigate three scenarios to quantify the potential extent and location of future oil palm expansion and resulting net carbon emissions. The three scenarios are:

1. *Business as Usual (BAU)*: This assumes that current corporate practices and state policies in Indonesia, Malaysia and PNG will remain in place. The projected rate of oil palm expansion is based on historical rates over the period 2005-2010 and is maintained constant in absolute terms (i.e., hectares, not %) for the first two decades of the simulation period to reflect projected global demand, but then declines as the human population is assumed to stabilize by 2050. Yields are assumed to remain at 3.7 Mg ha^{-1} of crude palm oil, the approximate mean yield of the industry in 2010 for the region under study.
2. *Moratorium on Peat (MRT)*: Rates of expansion are lower than the BAU scenario based on the assumption that global demand will be met partially by increased levels of productivity of 0.7% annually for a total yield increase of 25% by 2050, a conservative estimate of the potential yield increase of oil palm (Basri *et al*., 2005). The MRT scenario assumes that current corporate practices and state policies in Indonesia, Malaysia and Papua New Guinea are modified to halt expansion after 2010 on peat and high biomass landscapes (i.e., those with above-ground carbon stocks greater than 40 Mg ha^{-1}). Expansion continues, but is displaced to landscapes on mineral soils with low biomass, including disturbed forests, shrub, and grassland, as well as agricultural lands, other types of plantations and agroforest landscapes.
3. *Restoration of Peat (RET)*: Rates of expansion and productivity increases are identical to the MRT scenario and it is likewise assumed that corporate practices and state policies are modified to halt all expansion on peat and high biomass landscapes (i.e., above-ground carbon stocks greater than 40 Mg ha^{-1}) after 2010. However, the RET scenario also assumes that plantations on peat that exist in 2010 are gradually retired, rewetted, and restored to natural forest. The process of restoration is assumed to start in 2020 and proceeds gradually at an annual rate of 4% of all plantations on peat over the next 25 years. To compensate for the lost production of retired peat plantations, expansion on landscapes with mineral soils and low biomass are proportionally increased so that total rates of expansion and total production area remain the same as in the MRT scenario.

MODEL INPUTS AND ASSUMPTIONS

We modeled the three scenarios of future palm oil expansion and evaluated associated impacts on carbon emissions for Kalimantan, Sumatra, Papua (Indonesia), Sarawak and Sabah (Malaysia), and Papua New Guinea[1]. Land cover and land cover change maps for each region were developed by Gunarso *et al.* (2013 – this publication) and provided the input land cover data used in this analysis. These authors documented historical land cover change associated with the expansion of industrial scale oil palm plantations by visually interpreting Landsat satellite images and creating a region-wide map with 22 different land cover types spanning three temporal epochs (1990 to 2000, 2001 to 2005 and 2006 to 2009/2010). For each region, we used these maps to estimate historic rates of oil palm expansion into different land cover categories and extrapolated these rates of expansion into the future using simple regression models. We simulated future locations of expansion using spatial modeling techniques described below. Under the two alternative scenarios, overall rates of expansion were lower than the BAU scenario, and we varied our assumptions about where expansion was allowed to occur based on hypothetical policies and programs to promote climate change mitigation.

Historical and Projected Rates of Oil Palm Expansion

Figure 1 illustrates the assumptions that serve as the basis of the modeling scenarios. Curves correspond to the total area of historical and projected oil palm development per region under the BAU and the two alternative scenarios (MRT and RET). Historical data from 1990 to 2010 are based on Landsat imagery interpretation by Gunarso *et al.* (2013 - this publication). Projections beyond 2010 were made to

[1] Peninsular Malaysia was not included in our analysis as future expansion of palm oil is expected to cease in this region due to limited land availability.

reflect anticipated global demand and are described below (Figure 1):

A) In Sumatra, the rate of oil palm expansion tracks the 2005-2010 period, but then declines to an asymptote and remains flat thereafter;

B) In Kalimantan, the historical exponential rise of oil palm over the 2005-2010 period transitions into linear growth and declines to an asymptote by 2050;

C) In Papua, the rate of oil palm expansion increases over time, reflecting Indonesia's commitment to double production by the year 2020. The projection for Papua was determined by subtracting projected expansion in Sumatra and Kalimantan from total projected expansion across Indonesia, essentially transferring the past growth from Sumatra, Kalimantan and other regions of Indonesia, to Papua;

D) In Sarawak, oil palm expansion tracks the 2005-2010 period and reflects the state government's commitment to continue expansion and development;

E) In Sabah, oil palm expansion declines after 2010 and the area under plantation remains flat thereafter;

F) In Papua New Guinea, palm oil expansion increases through time, starting at the rates documented for 2005-2010, but increases exponentially thereafter reflecting the Papua New Guinea government's licensing of 5 Mha of oil palm concessions.

Simple regression models constructed to correspond to each curve, shown in Figure 1, were incorporated into the spatial modeling process.

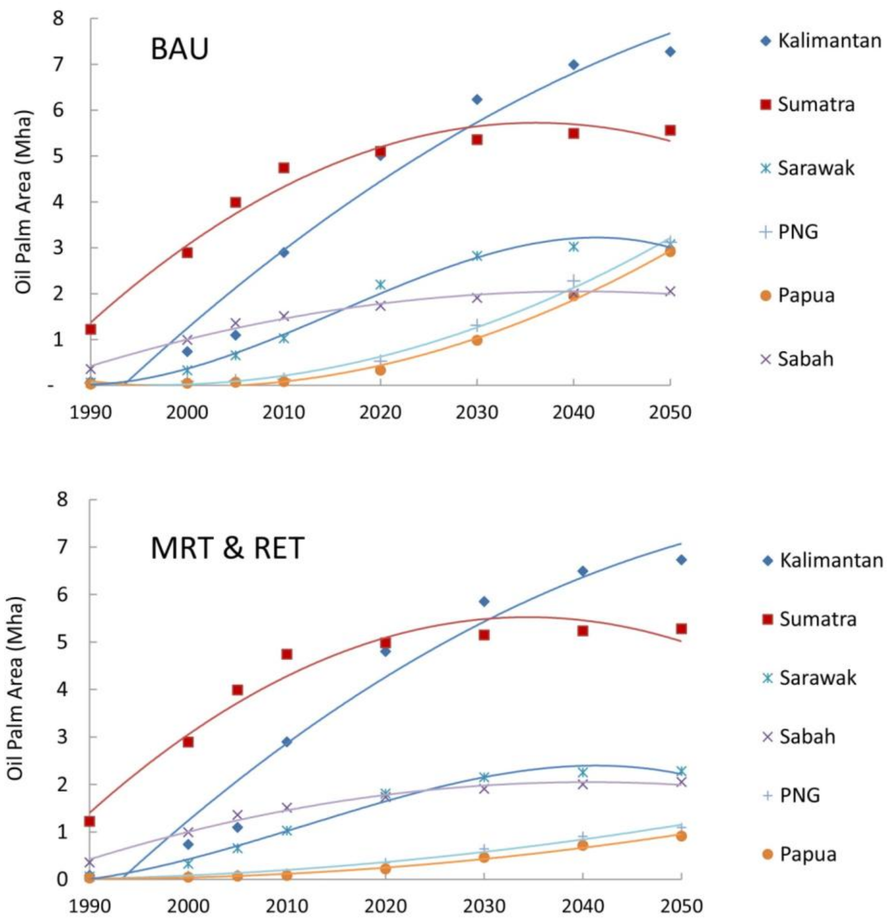

Figure 1. Projected expansion of oil palm plantations by region: a) a Business As Usual (BAU) and b) two alternative scenarios that assume a moratorium on new plantations on peat and high biomass landscapes (MRT) and a similar scenario that assumes total plantation area and rates of change remain the same, but that existing plantations on peat are rewetted and restored to peat forest after one 25-year cycle of production (RET). Points up to the year 2010 are empirical, points after 2010 are projected.

Simulation of Future Oil Palm Expansion

After rates of anticipated oil palm expansion were established under each scenario (Figure 1), we used the spatial modeling software GEOMOD to develop predictions of where this expansion was most likely to occur in each region. GEOMOD is a data driven, spatially explicit simulation model of land cover change that uses maps of ecological and socioeconomic attributes and of existing land cover to extrapolate the pattern of land use over time (Pontius *et al.,* 2001; Schneider & Pontius, 2001). The goal of GEOMOD is to generate potential scenarios of land cover change and evaluate which scenario has the best correlation to actual land cover change. The best scenario is then used to predict land cover change either forwards or backwards in time. GEOMOD has been used to model land cover changes across the world in many different ecosystems such as in Costa Rica (Pontius *et al.,* 2001), Indonesia (Harris *et al.,* 2008), India (Rashmi & Lele, 2010), the Mediterranean (Geri *et al.,* 2011), Massachusetts, USA (Schneider & Gil Pontius, 2001), Puerto Rico (Murphy *et al.,* 2010), Chile (Echeverria *et al.,* 2008), and the Hudson River Basin, USA (Hong *et al.,* 2011). Pontius *et al.,* (2001) give the most complete peer-reviewed description and application of GEOMOD.

Data Inputs

In its simplest form, GEOMOD requires three inputs. The first is a raster map depicting the current extent of the land-cover class of interest (e.g. oil palm), other areas available for conversion (e.g., forests, agriculture), and areas unavailable for conversion (e.g., open water, protected areas, etc.). The second input is a suitability map (or 'threat map', if we assume that areas highly suitable for development are under a high threat of conversion); this can be provided externally or can be created from user-selected factor maps as a prelude to the modeling exercise. Finally, the user must specify the magnitude (quantity) of expected change, either into or away from the land-cover class of interest.

For this analysis, land cover maps developed by Gunarso *et al.,* (2013 – this publication) for the years 2000, 2005 and 2009/2010 were used as input data to indicate locations of historical conversion of other land cover types to oil palm. Peat soils were defined for Kalimantan and Sumatra using a peat depth map from Wetlands International (Wahyunto & Suryadiputra, 2008). For remaining sites, peat soils were defined as histosols and fluvisols as in the Harmonized World Soil Database. Fluvisols aren't true peat soils, but the observed spatial correlation between fluvisols and the other peat layers on geographies where they overlapped suggested confusion between these types in the classification process. By including fluvisols, our modeled MRT and RET scenarios are more conservative regarding future development on peat.

For each region, we created the suitability map required for model simulation by combining various maps which spatially portray geographic attributes that may potentially influence the expansion of oil palm, either by contributing to or constraining its expansion (Table 2). The creation and selection of the suitability maps used in this analysis are described in further detail below and in Appendix 1. The quantity of expected change was specified based on the oil palm expansion curves shown in Figure 1.

Suitability Map Selection

Simulation of locations of oil palm expansion into the future is based on a map of suitability that indicates where this expansion is allowed to occur in the model. After several candidate maps were created from different combinations of factor maps (Appendix 1), one map was chosen to be used for simulation of future expansion based on its Relative Operating Characteristic (ROC).

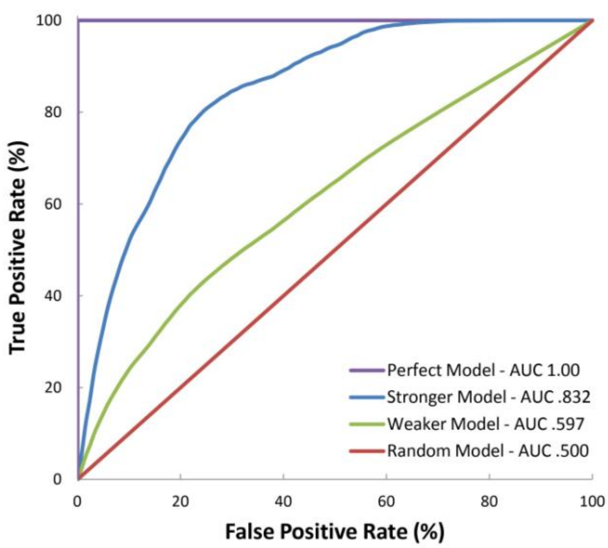

Figure 2. Examples of the Relative Operating Characteristic (ROC) used to evaluate candidate suitability maps in GEOMOD; AUC is Area Under Curve.

The ROC results from a comparison of each candidate suitability map with a reference map of actual oil palm expansion over a historical time period. To define the ROC, the rate of true-positives (change in both reality and model) is plotted on the vertical axis and the rate of false-positives (change in model, no change in reality) on the horizontal axis (Figure 2). The ROC statistic is calculated as the area under the curve (AUC). If the sequence of the suitability values matches perfectly the sequence in which real land-cover change has occurred, then the ROC would be equal to 1, because as the amount of modeled change increases from 0% to 100% (horizontal axis in Figure 2), there are no (0%) false positives and all (100%) are true positives (purple line in Figure 2). If the order of suitability values were assigned at random locations across the landscape, then the expected value of the ROC would be 0.5, as shown by the red line in Figure 2.

Simulation of Future Expansion

Given the three data inputs of a starting land cover map, a suitability map that indicates where change can occur, and a rate of expected change, GEOMOD follows a simple algorithm to simulate future land cover change from 2010 to 2050. GEOMOD assigns change one pixel at a time within the available area of the suitability map, starting with the cell with the highest suitability value. GEOMOD repeats this process, progressively assigning change to the pixel with the highest suitability until the model has reached the total amount of change specified by the user.

To ensure that simulated future expansion was consistent with actual past expansion with respect to land cover type, we stratified expected rates of regional land cover change. In other words, in addition to specifying within GEOMOD that 100,000 ha of expansion between 2010 and 2020 will occur within a region, we also defined how much of this change in area should occur for each land cover type and how much should occur on peat vs. mineral soils. This was done because land cover type significantly influences the magnitude of resulting carbon emissions. In all scenarios, oil palm expansion was excluded in protected areas, areas of urban development, water bodies, and areas unlikely to undergo conversion such as mangrove swamps, aquacultures and mines. For the MRT and RET scenarios, an additional constraint of peat soils was included.

Because swamp classes in the land cover maps and areas identified as peat in the soil type map used in developing the suitability map did not overlap exactly, we defined peat as the union of the swamp land-cover classes (swamp forests, open swamp) and areas identified as peat for Kalimantan and Sumatra or as histosols and fluvisols on the soil type map for other regions (see above). In the MRT and RET scenarios, we also added a factor into the suitability map that identified low biomass areas as more suitable than high biomass areas. To do this, we modified suitability scores using a map of above ground biomass (Saatchi *et al.,* 2011). This modified score was derived using the following equation:

$$S_b = \left(\frac{2}{3} \times \frac{S}{S_{max}}\right) + \left(\frac{1}{3} \times \frac{AGB_{max} - AGB}{AGB_{max}}\right)$$

S	Original suitability scores from the BAU scenario
S_{max}	Maximum suitability score for region
AGB	Above ground biomass (Mg/ha)
AGB_{max}	Maximum above ground biomass value for region
S_b	Modified suitability score used in the MRT and RET scenarios

In other words, the modified suitability score was weighted such that the non-biomass factors accounted for 2/3 of the final modified score and the biomass factor accounted for 1/3 of the final modified score. The weighting values were chosen through trial and error so that the biomass factor influenced the modeling outcome without overpowering the other factors that contributed to the analysis. The RET scenario used all the same constraints as the MRT scenario, but existing oil palm on peat as of 2010 was retired gradually over 25 years starting in the year 2020 and displaced to other, non-peat land cover types. The oil palm plantations that were retired were assumed to be restored to peat forest. Assumptions used to constrain expansion into specific land cover types are given in Table 1.

Table 1. The assumed percentage of oil palm area expanded into each land cover class in each region under the business as usual (BAU), moratorium on peat (MRT and restoration of peat (RET) scenarios.

Land Cover Class		Sumatra	Kalimantan	Papua	Sabah	Sarawak	PNG
Upland Forest (UDF+DF)	BAU	4	34	75	76	24	57
	MRT	0	15	60	0	0	40
	RET	0	15	60	0	0	40
Shrub/Grasslands (SCH+GRS)	BAU	1	38	1	0	1	0
	MRT	30	55	30	34	34	40
	RET	34	55	30	34	34	40
Swamp Forest (USF+DSF)	BAU	15	10	4	1	16	0
	MRT	0	0	0	0	0	0
	RET	0	0	0	0	0	0
Open Swamp (SSH+SG)	BAU	17	11	2	1	0	8
	MRT	0	0	0	0	0	0
	RET	0	0	0	0	0	0
Agroforestry (MTC+TPL+RPL/CPL)	BAU	43	0	14	8	37	0
	MRT	33	15	5	33	33	10
	RET	33	15	5	33	33	10
Agriculture (DCL+RCL)	BAU	7	5	0	0	0	25
	MRT	33	15	5	33	33	10
	RET	33	15	5	33	33	10
Remainder (Mainly BRL)	BAU	14	3	4	14	22	10
	MRT	0	0	0	0	0	0
	RET	0	0	0	0	0	0

In summary, differences among scenarios were operationalized in three primary ways: 1) Net expansion projected for each region was scenario dependent, with MRT and RET scenarios experiencing lower net expansion; 2) The proportion of oil palm expansion allocated to each land cover class varied by scenario, with the MRT and RET scenarios receiving no allocation on peat and less allocation in high-biomass upland forest (see Table 1); and 3) in the MRT and RET scenarios, suitability for expansion was modulated by modifying the suitability map to incorporate a biomass factor, with lower suitability scores assigned to pixels with higher biomass.

Model Validation

The model was trained using data from the 2000 to 2005 period and validated based on a comparison between simulated land cover change and actual land cover change for the time period 2006 to 2010. As described above, the ROC statistic compares candidate suitability maps against locations of oil palm across all hypothetical quantities of simulated change and is used to select the suitability map that most accurately predicts historical change. In contrast, the Figure of Merit (FOM) statistic evaluates the model's performance in accurately simulating specific locations of land cover change as a function of the quantity of change that has actually occurred on the landscape. A model that has a high ROC score usually translates into a model with a high FOM score, as long as simulated land cover change follows the logic that the most suitable areas are converted first. If actual land cover change occurred in highly unsuitable areas for whatever reason, then the ROC score could be high while the FOM score could be low.

The FOM is calculated as the ratio of the amount of overlap between observed and predicted change to the union of observed and predicted change (Figure 3). In other words, it is the agreement between the *actual* change that occurred between time 1 and 2 according to the reference maps, and the *simulated* change predicted to occur between time 1 and 2. The FOM ranges from 0%, where there is no overlap between observed and predicted change, to 100% where there is perfect overlap between observed and predicted change. These theoretical values ranging between 0 and 100% are potentially deceiving, however, because as the proportion of area to be converted increases, so too does the probability of a correct prediction due simply to chance.

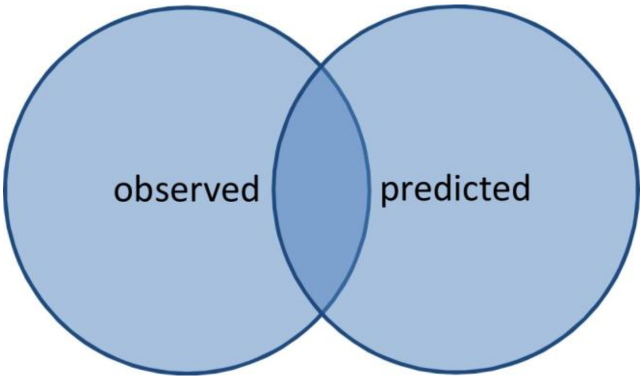

Figure 3. Schematic of how the Figure of Merit statistic is calculated as the ratio of the amount of overlap between observed and predicted change (darker blue shaded region) to the union of observed and predicted change (all shaded regions).

Quantification of Net Carbon Emissions Associated with Oil Palm Expansion

The potential net carbon emissions resulting from biomass clearing for the expansion and operation of oil palm plantations were estimated for each scenario by overlaying the predicted land cover change maps produced by GEOMOD with estimates of biomass for each land cover type converted. For land cover classes dominated by woody vegetation, we estimated biomass using pixel-based estimates from Saatchi *et al.* (2011) which were derived from remote sensing calibrated by forest inventory data. For agriculture, oil palm and agroforest classes, we used average biomass values derived by Agus *et al.* (2013a – this publication). In locations of simulated conversion to oil palm, net carbon emissions from biomass were estimated as the difference between the carbon stock in the original land cover type and the time-averaged carbon stock of the oil palm plantations (36 Mg C ha^{-1}). Although outputs from GEOMOD are simulated maps of specific pixels projected to be converted to oil palm in the future, we acknowledge the limitations of GEOMOD's ability to predict expansion at this level of spatial precision. Therefore, when estimating carbon emissions from simulated oil palm expansion, we use the Saatchi *et al*. (2011) biomass map to estimate an average carbon stock value per land cover stratum, and combine these values with the area converted per stratum. In this way, our emissions estimates are less sensitive to the model's ability to correctly predict the exact pixel to be converted and therefore less sensitive to the pixel level variability of carbon stock values.

Carbon emissions from below ground biomass and soil organic matter from land cover types located on mineral soils were omitted from our analysis due to insufficient data, but emissions from peat drainage for oil palm cultivation were estimated by assuming an emission factor of 43 Mg CO_2 ha^{-1} yr^{-1} (Agus *et al.,* 2013a – this publication). Once an area was converted to oil palm, peat emissions were assumed to occur in the year of drainage and every year thereafter over the simulation period. A one meter layer of peat stores 1100-2600 Mg CO_2 ha^{-1} (Page *et al*., 2002; Agus & Subiksa, 2008); therefore we assumed that the available peat supply for oxidation would continue throughout the simulation period. We did not include emissions from peat fires in the analysis, because it is illegal to use fire as a management tool and we assumed that all future plantations will be legally complaint. We assumed that the RET scenario includes displacement of existing plantations currently operating on peat soils to low biomass landscapes located on mineral soils. We also assumed that once oil palm plantations on peat are taken out of production, the landscape would be rewetted, which would halt emissions from the oxidation of peat and lead to carbon sequestration in the above ground biomass pool of re-growing forest vegetation. Upon reforestation, biomass was assumed to accumulate at rates specified for tropical rain forest by

IPCC (2006)[2]. Biomass was converted to carbon using a conversion factor of 0.5 (IPCC, 2003).

RESULTS

Identification of High Threat Areas for Oil Palm Expansion and 'Sustainable' Alternatives

Spatial factors included in the final suitability maps for oil palm development for each region are shown in Table 2 and include: elevation, slope, soil type, proximity to roads and to urban areas, as well as the presence of palm oil mills and oil palm concessions (see Appendix 1). However, the final 'suitability' maps (Figure 4) are based only on spatial factors correlated with historical expansion and do not reflect 'suitable' areas for development with respect to sustainability criteria, such as locations on peat or in high biomass areas. We therefore reclassified the maps to develop a visual assessment of where the 'threat' of oil palm expansion was highest, both on peat soils (Figure 5) and in high biomass landscapes (Figure 6). Each map portrays three categories of low, medium and high risk of conversion to oil palm. For all sites, high risk was defined as pixels in the 90th percentile of suitability, medium as pixels in the 80th-90th percentile, and low as pixels below the 80th percentile.

We also quantified the extent of land in each region available for 'sustainable' expansion, which we defined as landscapes below 1000 m elevation on mineral soils and with above ground biomass carbon values below 40 Mg ha^{-1} that were located outside of protected and excluded areas.

The availability of these lands appears to be highest in Sumatra and lowest in Sarawak and Sabah (Table 3). In Sarawak, the amount of land required for oil palm expansion by 2020 under all three scenarios exceeds the available supply of so-called 'sustainable' land. For all other regions except Sumatra, the area needed to accommodate rates of oil palm expansion under the BAU scenario will either come close to or exceed the amount of available sustainable land by 2050. In Sumatra, there is an abundance of previously deforested land with low to moderate above ground biomass situated on mineral soils at elevations below 1000 m. In the case of Papua and Papua New Guinea, where expansion by 2020 may be achievable sustainably, expansion beyond 2020 will exceed the limits of available supply of sustainable land. Moreover, although the amount of land that meets our criteria for sustainable expansion is sufficient to accommodate desired expansion over the medium term, these areas are not contiguous and represent a patchy distribution of available land (Figure 7). Consequently, the amount of land available may be sufficient on the island of New Guinea, but the spatial distribution of suitable areas is not conducive to the development of spatially contiguous plantation estates.

[2] Above ground biomass growth rate is 13 Mg dry matter ha^{-1} yr^{-1} for forests <20 years, and 3.4 Mg dry matter ha^{-1} yr^{-1} for forests >20 years (see Table 4.9, IPCC Guidelines for AFOLU, 2006).

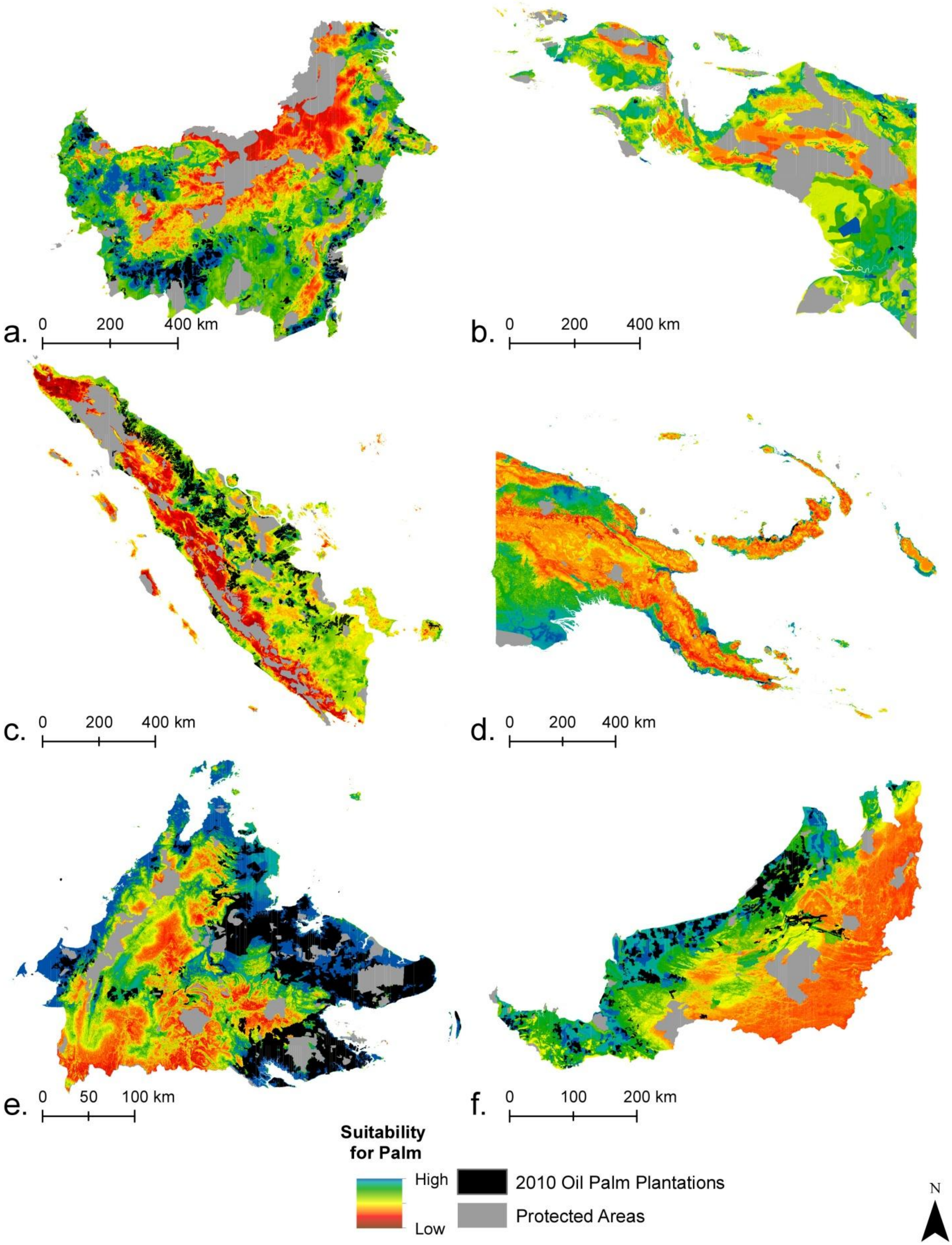

Figure 4. Suitability maps for oil palm expansion by region. (a) Kalimantan, Indonesia; (b) Papua, Indonesia; (c) Sumatra, Indonesia; (d) Papua New Guinea, (e) Sabah, Malaysia; (f) Sarawak, Malaysia.

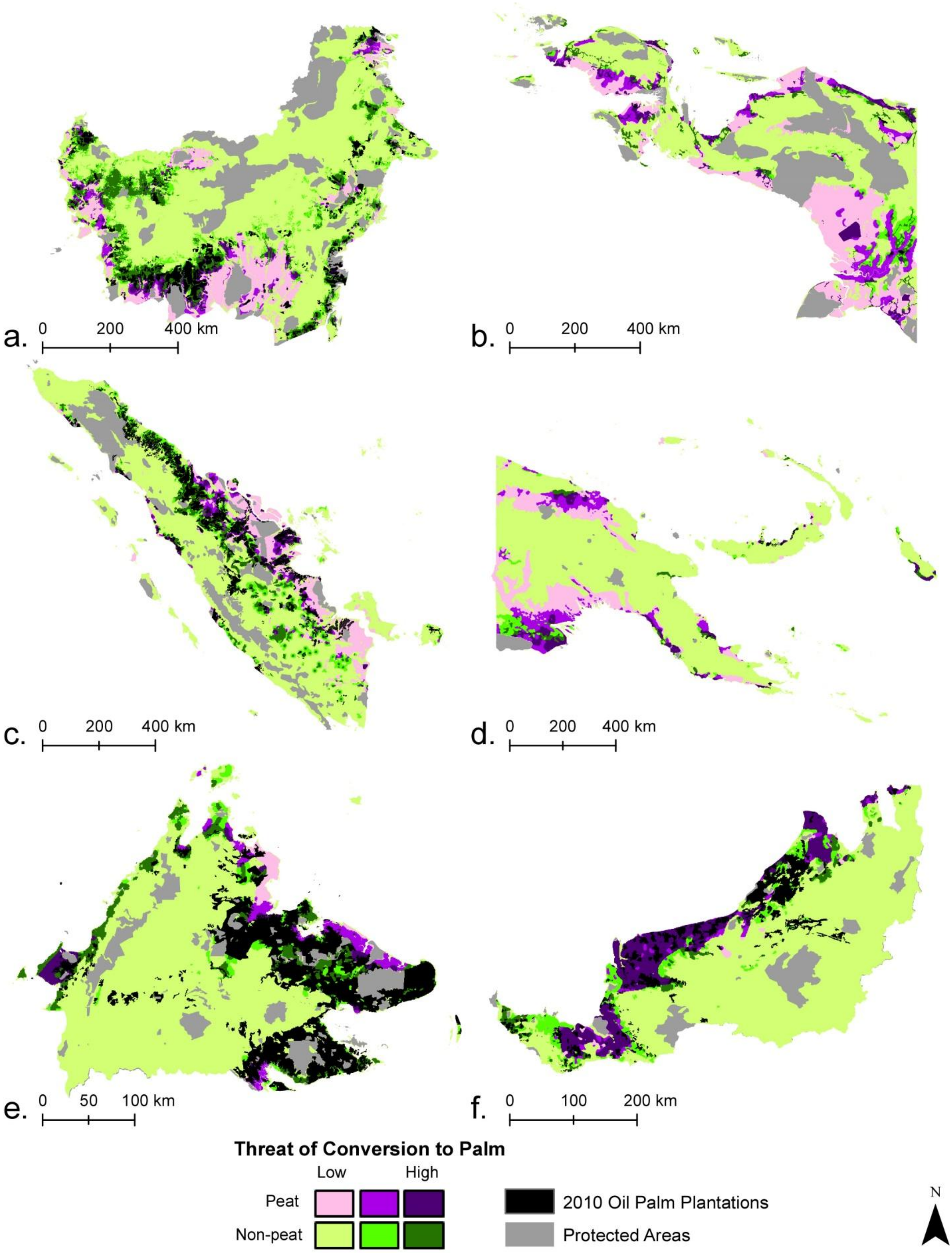

Figure 5. Threat maps for palm oil expansion into peat by region. (a) Kalimantan, Indonesia; (b) Papua, Indonesia; (c) Sumatra, Indonesia; (d) Papua New Guinea, (e) Sabah, Malaysia; (f) Sarawak, Malaysia.

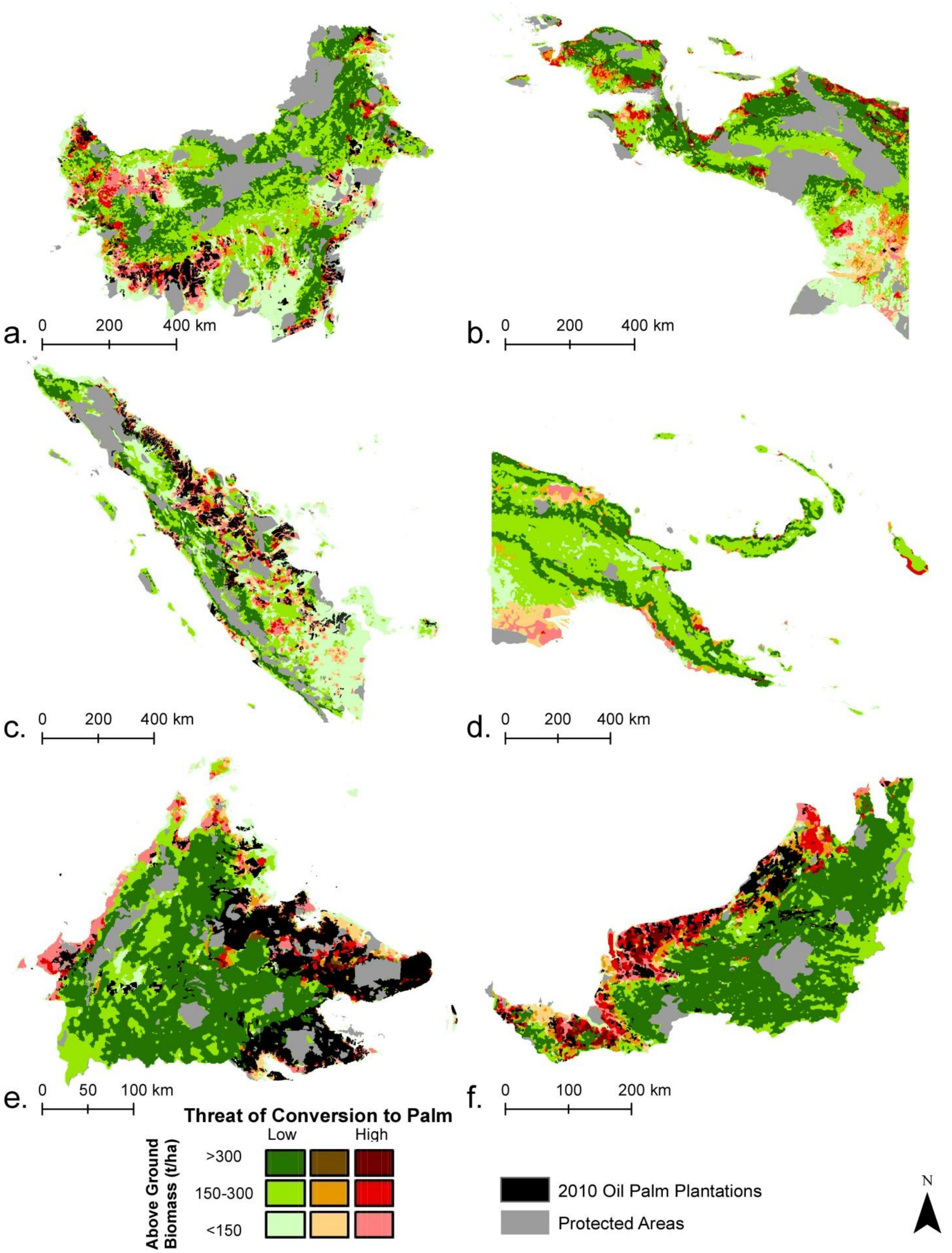

Figure 6. Threat maps for palm oil expansion into high biomass areas (regardless of soil type) by region. (a) Kalimantan, Indonesia; (b) Papua, Indonesia; (c) Sumatra, Indonesia; (d) Papua New Guinea, (e) Sabah, Malaysia; (f) Sarawak, Malaysia.

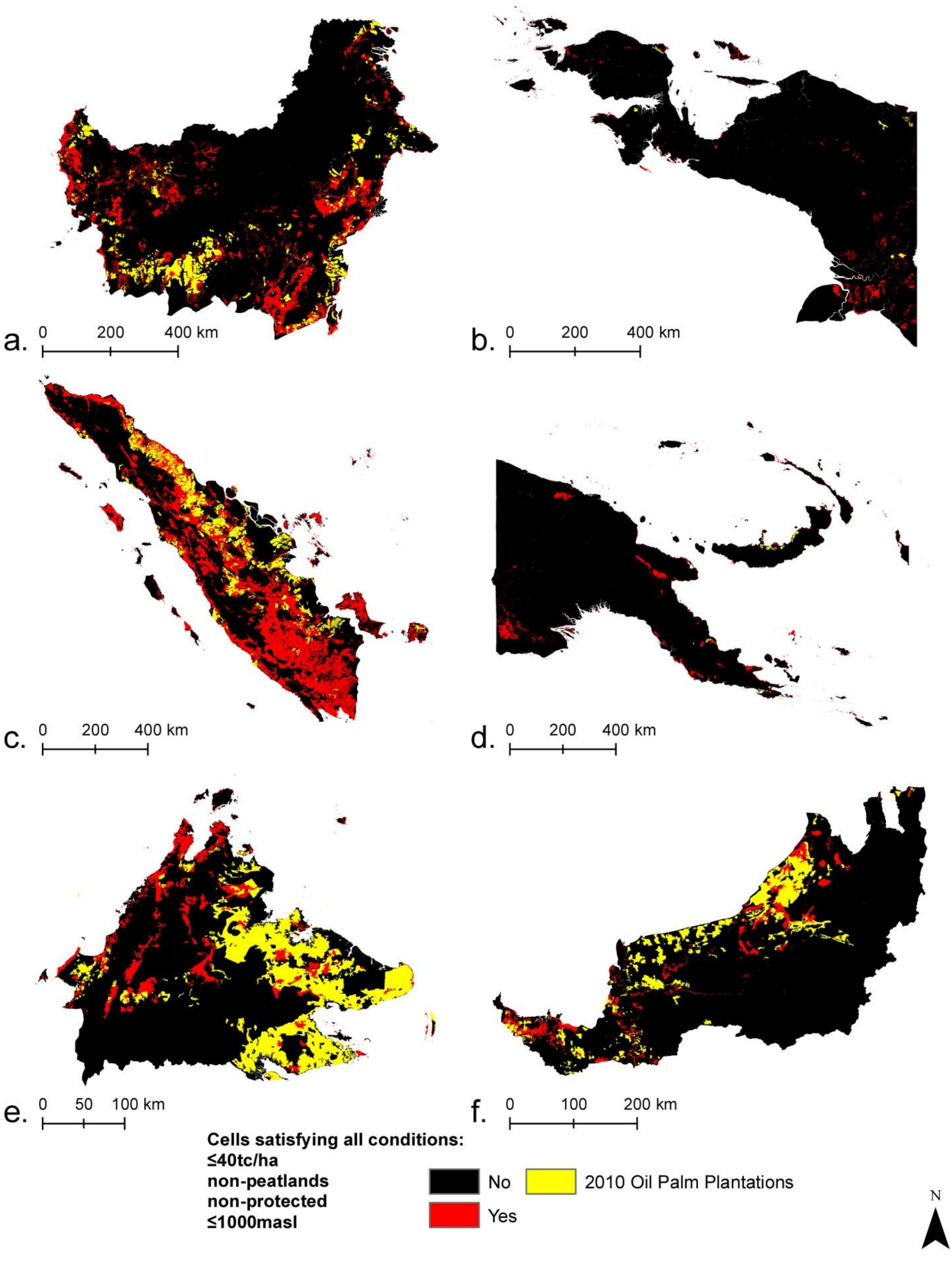

Figure 7. Areas for 'sustainable' palm oil expansion per region where 'sustainable' is defined as biomass carbon stocks less than or equal to 50 Mg C ha^{-1}, and exclude peat, protected and other excluded areas and which are below 1000 m elevation. (a) Kalimantan, Indonesia; (b) Papua, Indonesia; (c) Sumatra, Indonesia; (d) Papua New Guinea, (e) Sabah, Malaysia; (f) Sarawak, Malaysia.

Table 2. Spatial factors used for projecting oil palm expansion in the study regions.

Region	Elevation	Slope	Roads	Palm Oil Mills	Concessions	Urban Areas	Soil Type
Papua	x	x	x		x		x
Kalimantan	x	x	x	x	x		x
Sumatra	x	x	x	x	x		x
Sabah	x	x	x				x
Sarawak	x	x	x		x		x
PNG	x	x	x			x	x

Table 3. The geographic area available for sustainable oil palm expansion in 2010, 2020 and 2050 compared to the area required to be developed under each scenario. Sustainable land refers to areas on mineral soils (non-peat), with above ground carbon stocks < 40 Mg C ha^{-1}, below 1000 m elevation, and not under protection. (Open water, mining, and other non arable lands are excluded.)

Region	Total available in 2010	Area needed for new expansion (10^3 ha)					
		2020			2050		
		BAU	MRT	RET	BAU	MRT	RET
Kalimantan	5,307	2,267	2,055	2,092	4,529	3,985	4,450
Papua	904	247	137	141	2,833	830	871
Sumatra	15,700	375	256	376	836	553	2,058
Sabah	632	247	247	259	564	564	717
Sarawak	571	1,173	783	816	2,053	1,254	1,667
PNG	1,463	395	220	222	2,988	960	982
TOTAL	24,576	4,704	3,698	3,906	13,802	8,147	10,744

Table 4. Percentages of total net carbon emissions attributable to each region under three scenarios (BAU, MRT and RET) by the years 2010, 2020 and 2050.

Region	Historical	BAU		MRT		RET	
	by 2010	by 2020	by 2050	by 2020	by 2050	by 2020	by 2050
Kalimantan	39	34	23	35	28	36	30
Papua	1	4	15	3	5	3	6
Sumatra	38	31	23	40	43	41	37
Sabah	8	7	4	5	5	5	4
Sarawak	13	20	21	14	14	12	17
PNG	1	4	13	3	5	3	6

PROJECTED EMISSION ESTIMATES

In the BAU scenario, total cumulative CO_2 emissions were projected to exceed 15 Pg CO_2 by 2050 (i.e. 15 Gigatons), while mean annual emissions increased from 264 Tg CO_2 yr^{-1} between 2010 and 2020 to more than 424 Tg CO_2 yr^{-1} between 2040 and 2050 (Figure 8). The increase in future emissions is due to further emissions from above ground biomass due to land use change (LUC), as well as continuous and incremental emissions from the oxidation of peat drained for oil palm plantations. Between 2000 and 2010, Kalimantan was the largest source of total emissions, followed by Sumatra, which together accounted for 76% of total net cumulative emissions in 2010. However, as the oil palm industry continues to expand into new regions of Papua, Sarawak and Papua New Guinea, these regions eventually have similarly large emission profiles (Table 4). Sarawak, which by 2050 will have converted most of its peat soils to oil palm, actually displaces Kalimantan as the second largest source of emissions when calculated on an annual basis (Figure 9).

In the MRT scenario, the presumed moratorium on oil palm expansion on peat soils and the exclusion of development on high biomass landscapes on mineral soils cause total cumulative emissions to decrease dramatically to 6.4 Pg CO_2 by 2050. Mean annual emissions are projected to drop below historical values, falling from 165 Tg CO_2 yr^{-} between 2000 and 2010 to less than 134 Tg CO_2 yr^{-1} by 2050. The lower emission profile is the consequence of the stabilization of emissions due to peat oxidation (113 Tg CO_2 yr^{-1}) and the gradual reduction of emissions from above-ground biomass (31 to 11 Tg CO_2 yr^{-1}) as LUC also falls over time, due to yield increases and the reduced demand for land (Figure1). Sumatra is the largest source of emissions both in cumulative or annual terms (Table 4), and although new expansion does not occur on peat soils, the ongoing drainage of existing oil palm plantations is a continual source of emissions (Figure 9).

In the RET scenario, our model assumes that 4% of all existing oil palm plantations are retired starting in 2020 and displaced to more sustainable areas over the following 25 years. As plantations are decommissioned, the model assumes they are rewetted and restored to natural forest, thus becoming a carbon sink rather than a source of CO_2 emissions. The total area under cultivation remains the same, however, and an equivalent area of new oil palm plantations are established on mineral soils. The total cumulative emissions that might result from these assumptions are 5 Pg CO_2, which are one third of those produced in the BAU scenario by 2050 and lead to a cumulative emission reduction of 10 Pg CO_2 between 2020 and 2050. When the RET scenario is compared to the MRT scenario, the restoration of peat would lead to an additional net cumulative reduction of 1.78 Pg CO_2 between 2020 and 2050. Total mean annual emissions fall from a high of 157 Tg CO_2 yr^{-1} in 2020 to only 12.5 Tg CO_2 yr^{-1} by 2045, remaining constant thereafter due to what are essentially residual levels of LUC on mineral soils due to a shortage of land which meets the defined criteria for sustainable use in Sabah and Sarawak.

In Sabah, it was not possible to simulate either the MRT or RET scenarios using our initial assumptions of equal allocation of new plantations into shrub/grass, agroforestry, plantations, and agriculture (Table 1) because the area of these land cover types needed to fulfill model assumptions exceeded the available supply. For the MRT scenario, we modified our assumptions and allocated future expansion into existing agricultural land, while under the RET scenario, the model was forced to allocate the relocation of oil palm plantations on peat to upland forest after the availability of other land cover types was exhausted. The situation was similar in Sarawak, where neither the MRT nor the RET scenarios could be simulated using our initial assumptions. The area of oil palm plantations on peat that needed to be shifted to other land cover types vastly exceeded the available supply and forced the model to allocate oil palm expansion to upland forested landscapes under both MRT and RET scenarios.

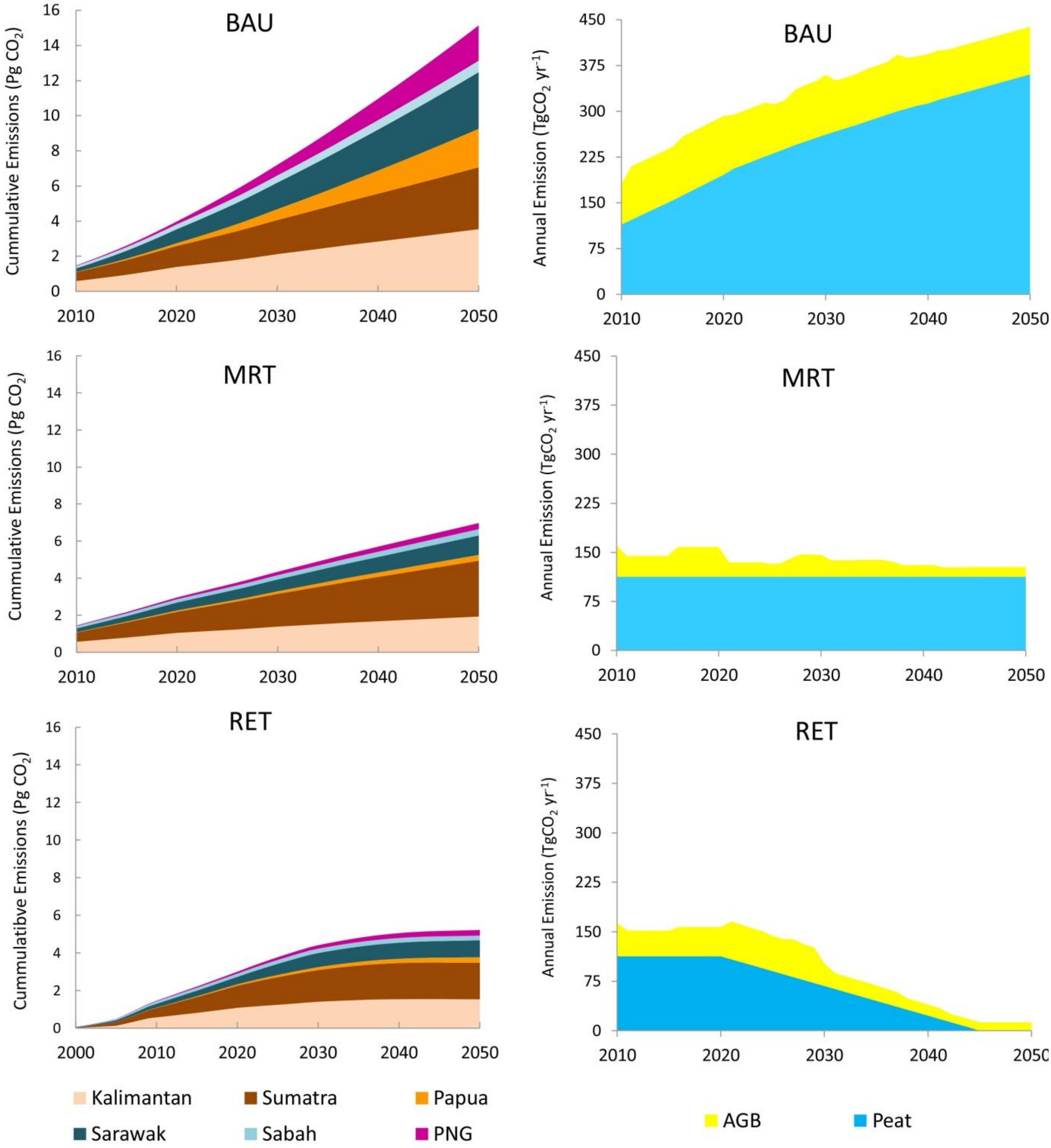

Figure 8. ***Left Column*****: Projected cumulative CO_2 emissions from oil palm expansion by region under three scenarios: Business as Usual (BAU), Moratorium on Peat (MRT), and Restoration of Peat (RET).** ***Right Column*****: Projected annual emissions summed over all regions for the same three scenarios.**

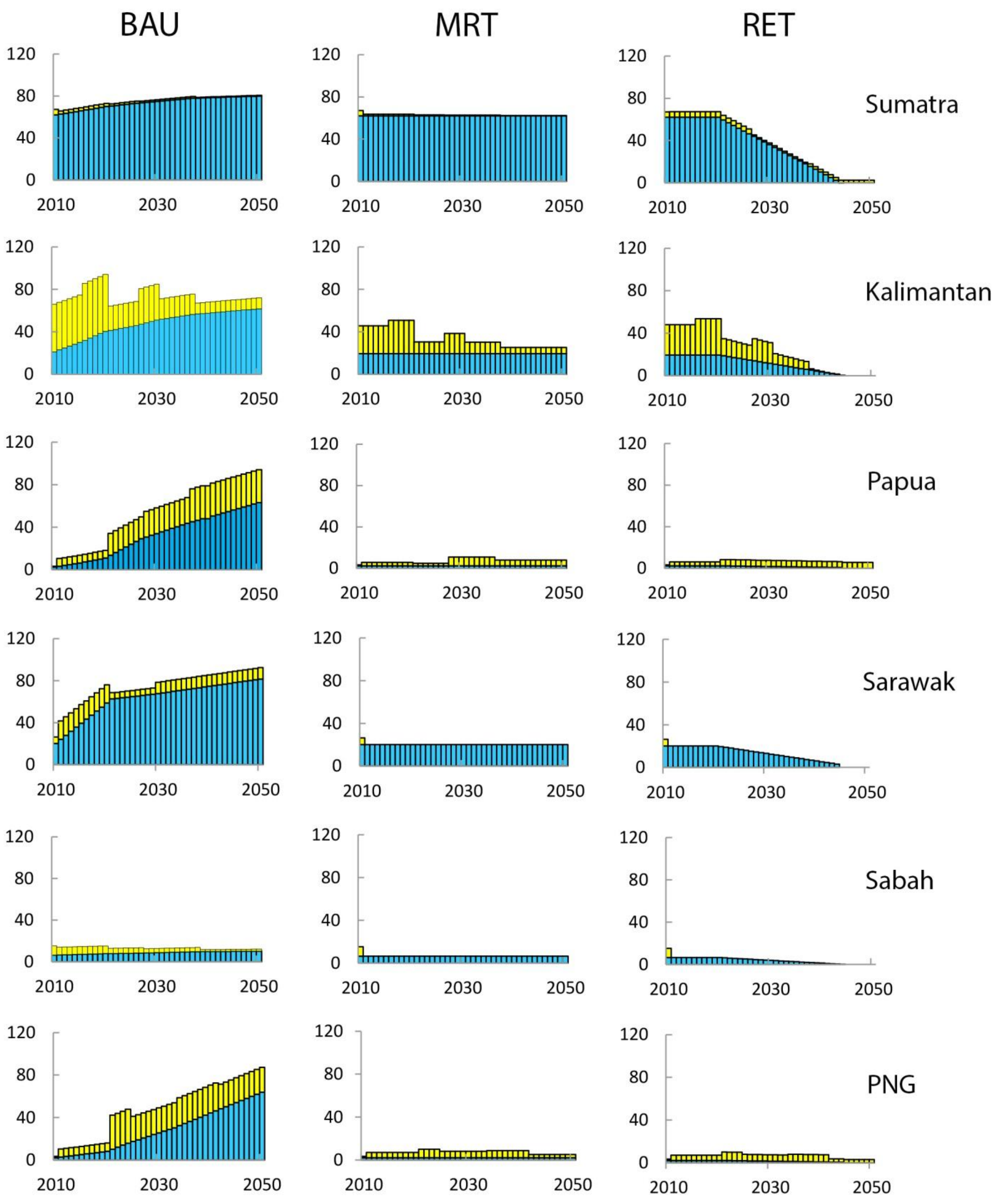

Figure 9. A graphical matrix of net annual emissions (Tg CO_2 yr^{-1}) over the simulation period (2010-2050): the rows of the matrix are regions and the columns are scenarios. Blue areas represent emissions from peat oxidation and yellow areas represent emissions from above-ground biomass due to land use change. The stepwise nature of the biomass missions are the result of non-linear curves (see Figure 1) that are represented as a series of linear time steps in the modelling process.

COMPARISON WITH HISTORICAL EMISSION ESTIMATES

A comparison of the emission estimates from our model using emissions factors based on mean carbon stock values derived from pixel-based estimates of above ground biomass (Saatchi et al., 2011) with the companion study that used mean carbon stock values derived from regional field based studies (Agus et al., 2013a, 2013b – this publication), showed substantial differences in the estimated emissions for the six regions covered by both studies, which included Sumatra, Kalimantan, Papua, Sarawak, Sabah and Papua New Guinea, but which excluded Peninsular Malaysia (Table 5). These differences, since they are derived from similar datasets, represent a source of uncertainty in the robustness of both historical and future emission estimates. The source of this variance is linked to the interpretation and selection of certain data inputs used for both modeling exercises: 1) different criteria for determining the spatial extent of peat formations and 2) different data sources used for calculating emission factors from changes in above-ground carbon due to LUC.

Specifically, we adopted a broad definition to define peat soils by using the 'union' of both the 'swamp forest' area as interpreted from Landsat images (Gunarso et al., 2013 – this publication) and the polygons based on soil maps (Wahyunto & Suryadiputra, 2008). In contrast, Agus et al. (2013b –this publication) used a narrow definition of peat swamp area based on peat areas delineated by soil scientists. Information on the extent and distribution of peat soils has been revised downward by some soil scientists (Ritung et al., 2011), and thus our model may overestimate emissions from this source.

Differences between emission factors linked to changes in above ground carbon due to land use change are linked to our decision to use a pixel-based pantropical dataset derived from remote sensing data calibrated by a pantropical network of forest plots (Saatchi et al., 2011). We selected this dataset in order to evaluate its utility for this type of analysis and to compare the outcome with the traditional approach of using regional field based forest plot studies (Agus et al., 2013a – this publication).

The results varied by region, but were generally similar for forest and swamp forest aggregate classes (Table 6), which suggest that the forest stratification employed by Gunarso et al. (2013 – this publication) is broadly accurate. This was not the case, however, for the two aggregate categories that circumscribe the shrub and grassland categories for both upland and swamp habitats. In those instances, the estimated mean carbon stock values derived from the pixel-based dataset were substantially higher than the estimates based on regional plot-based studies (Table 6). The higher mean carbon stock values derived from the pixel-based approach suggest that some of the areas identified as shrub land by Gunarso et al. (2013 – this publication) might be more appropriately classified as highly degraded forest rather than shrub land and the emission estimates due to LUC from Agus et al., (2013b – this publication) may under estimate total CO2 emissions from above ground carbon due to land use change.

Table 5. Comparisons of estimated historical carbon emissions based on land use change modelled by pixel-based approach (Saatchi *et al*., 2011) with field-based estimates of Agus *et al*. (2013b – this publication) for Sumatra, Kalimantan, Papua, Sarawak, Sabah and Papua New Guinea.

	Agus *et al.*, 2013b 2000 – 2005 Tg CO2 $ha^{-1} yr^{-1}$	Saatchi *et al.*, 2011 2000 – 2005 Tg CO2 $ha^{-1} yr^{-1}$	Agus *et al.*, 2013b 2005 – 2010 Tg CO2 $ha^{-1} yr^{-1}$	Saatchi *et al.*, 2011 2005 – 2010 Tg CO2 $ha^{-1} yr^{-1}$
AGC due to LUC	29.11	22.48	53.55	97.24
Peat Oxidation	48.57	62.28	79.66	99.53
Total	77.69	84.76	133.21	196.77

Table 6. Above- ground carbon (AGC) stocks (Mg C ha^{-1}) used to estimate CO_2 emissions between 2010 and 2050. The data from Agus *et al.* (2013a,b – this publication) are derived from published reports, while Saatchi *et al.* (2011) is a pixel-based estimate of AGC at 1-km^2 spatial resolution; values shown represent the average of all 1-km pixels within a land cover class as defined by Gunarso *et al.* (2013 – this publication) for each region.

Land Cover Class	From Agus *et al.* (2013a, b)			Derived from Saatchi *et al.* (2011)					
	Weighted mean	Aggregate Land Cover Class	SE Asia	Sumatra	Kalimantan	Papua	Sabah	Sarawak	Papua New Guinea
Undisturbed Upland Forest	189	Upland Forest	150	143	141	136	148	150	133
Disturbed Upland Forest	104								
Upland Shrub land	30	Shrub and Grassland	27	96	86	59	43	36	88
Upland Grassland	3								
Undisturbed Swamp Forest	162	Swamp Forest	130	121	114	117	96	128	108
Disturbed Swamp Forest	84								
Swamp Shrub land	28	Open Swamp	26	61	56	53	51	33	60
Swamp Grassland	2								
Timber Plantation	44	Agroforest / Plantations	51	Same as aggregate class averages					
Mixed Tree Crops	54								
Rubber plantation	55								
Oil palm Plantations	36	Oil palm Plantations	36						
Bare Land	36	Bare Land	3						
Upland crops	11	Agriculture	11						

Table 7. Validation statistics (%'s) for each modeled oil palm expansion in each region. The FOM Ratio (Model/Random) indicates the degree to which the model out-performs a hypothetical random model.

	Observed change % of available surface 2005-2010 used for oil palm	% already developed as plantations	Relative Operating Characteristic (ROC)	Figure of Merit (FOM) for Model Outcome	Figure of Merit (FOM) Random Model	FOM Ratio: Model/Random
Papua	0.036	0.2	0.94	0.8	0.02	45.94
Papua New Guinea	0.095	0.2	0.91	1.7	0.05	35.67
Sumatra	1.725	9.8	0.71	3.4	0.87	3.85
Sabah	2.685	20.2	0.54	3.4	1.36	2.48
Sarawak	3.223	5.7	0.83	19.8	1.64	12.10
Kalimantan	4.308	2.4	0.83	12.5	2.20	5.67

MODEL VALIDATION

ROC scores indicate highest model performance in Papua and Papua New Guinea, moderate performance in Kalimantan and Sarawak, and relatively low model performance in Sabah and Sumatra (Figure 10). Model fit, as evaluated by the ratio of the model's actual FOM against the FOM of a hypothetical random model, was also highest in Papua and Papua New Guinea and lowest in Sumatra and Sabah (Table 7).

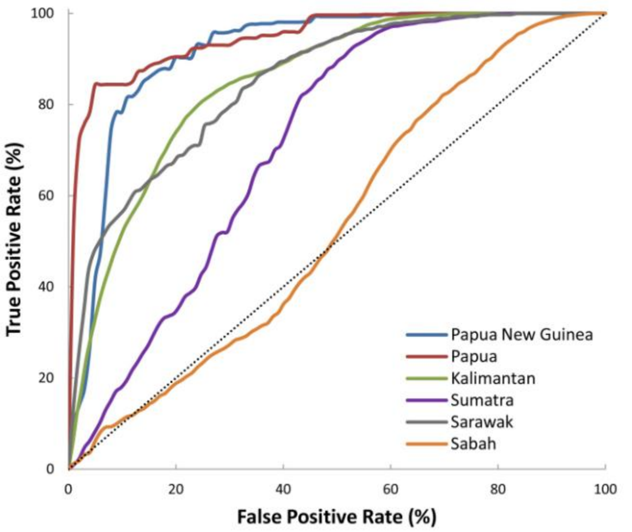

Figure 10. The final Relative Operating Characteristic (ROC) curves for each region's suitability map.

DISCUSSION

Model validation involves comparing the model's simulated map of land cover change to actual change. It is important to validate the model output cautiously, however, because a naïve interpretation can lead to misleading conclusions. For example, if 10% of the landscape actually changes a naïve model that predicts no change would still be 90% accurate. In other words, it is possible to obtain very high agreement in terms of percentages due simply to a large signal of persistence in the landscape. Taken together, the ROC and FOM statistics provide robust measures of model performance, which take into account persistence.

The raw FOM score for a given region is not particularly helpful for comparing how well the model performs across regions, because this statistic is sensitive to the amount of change within a particular region. As the amount of change increases, the probability of correctly predicting changed pixels also increases. If there is not much change, then the probability of predicting the exact location of where change will occur is low. Where the FOM is at least equivalent to the total area of change as a percentage of the total area, then the model is better than random. The higher the ratio between the model's FOM and the FOM of a random model, the better the model performs.

Results of validation for the six modeled regions can be categorized into three groups:

1) Papua and Papua New Guinea: Less-developed regions with low observed prevalence of oil palm. Spatial autocorrelation in palm development results in high ROC scores, while the low observed rate of change makes predicting exact pixels difficult and results in a low absolute FOM.
2) Kalimantan and Sarawak: Moderately-developed regions with a high rate of palm conversion. In these regions, ample highly suitable areas remained undeveloped in 2005, for which it is comparatively easy to accurately predict development, which results in high scores for both ROC and FOM.
3) Sumatra and Sabah: Highly-developed regions with moderate rates of change. In these regions, a large proportion of highly suitable areas have already been developed as oil palm plantations. Any further development is now on land with lower suitability scores, making prediction more difficult. Both ROC and FOM statistics are low in Sumatra and Sabah.

Papua New Guinea and Papua

The island of New Guinea is still largely covered by relatively pristine natural forest and represents a major frontier for future palm expansion. As of 2010, each of the two parts of the island contained only a small amount (< 150,000 ha) of oil palm plantation area, but the industry is expected to grow dramatically by 2050. In the BAU scenario, the extent of oil palm plantations would expand 35 fold in comparison to 2010 levels in Papua, and 23 fold in Papua New Guinea. Due to the rapid rate of expansion and the lack of an extensive history of palm development in these regions, uncertainty is high regarding future plantation distribution. While a high ROC score (> 0.9) suggests that these models are very strong, these results must be understood in the proper context. Natural spatial autocorrelation of values in driver variables, which is a common attribute of landscape variables, combined with a very small pool of calibration locations, means that a single large plantation can have a large impact on any indicators of model strength. While our model was able to predict well the distribution of oil palm locations in 2010, the anticipated high rate of future change makes it difficult to say with confidence that these scenarios adequately portray the spatial distribution of projected future expansion. In other words, a single new estate developed in a previously undeveloped area, which is actually highly likely, might radically change the model output. After a few years of future development have created a more extensive history of palm development, a more robust understanding of the drivers of change and the spatial distribution of oil palm expansion in Papua and Papua New Guinea is likely to emerge.

Kalimantan and Sarawak

These two frontier regions are anticipated to rapidly expand palm oil production over the next 40 years. However, unlike Papua and Papua New Guinea, these regions have already experienced a substantial amount of oil palm development. It is in these sites where the FOM and ROC metrics are most in balance. The ROC curve for Kalimantan shows that over 80% of observed change between the validation period of 2006 and 2010 occurred in the highest quartile of modeled suitability scores. In Sarawak, 75% of change occurred in this quartile. Because Kalimantan and Sarawak were in the early stages of expansion in 2005, there still remained large areas of the kind of low-lying and accessible undeveloped land that has typified past expansion. As long as there is a large supply of suitable land, then the model that uses these variables as drivers will do well at predicting future change accurately.

Sumatra and Sabah

These two regions are characterized by landscapes that had highly developed palm oil industries prior to 2000 and in our model were not expected to expand rapidly through 2050. In both of these regions, suitability maps based on the distribution of 2005 plantations predicted the observed palm expansion poorly over the validation period of 2005 to 2010. These regions contained the highest proportion of existing palm land cover in the 2000 calibration year, so it is notable that these suitability maps performed least well. Intuitively, a site with a large history of land-cover change should allow GEOMOD to effectively calibrate the influence of individual drivers.

One possible explanation for this discrepancy is that the decision making process regarding site selection exhibited between the validation period of 2005 and 2010 was fundamentally different than in earlier years. In validating the suitability map, GEOMOD bases its projections on the total palm distribution *ca.* 2005, a geographic distribution that represents the accumulated land conversion decisions of past years. If something has fundamentally changed in the prevailing logic of site selection among producers, then an empirically-derived suitability map based on past development will be a poor guide to future expansion.

In the case of Sumatra and Sabah, one quality shared between sites in our model is that they undergo a transition from rapid expansion to a more gradual, and eventually flat, growth profile. These findings suggest that the environmental and economic drivers that drove expansion between 1990 and 2005 are no longer as important relative to other variables because the highly suitable land in Sumatra and Sabah has already been converted and it is possible that producers are forced to weigh land use in a more nuanced way than during past expansion. In the case of Sumatra where there are large areas of low to moderate biomass landscapes (agroforest), the constraint on plantation development may be related to social criteria that we did not incorporate into our models, such as whether land is occupied or has land tenure regimes that are not conducive to plantation development. In Sabah, there is a limited amount of low to moderate biomass area on lowland landscapes that places a constraint on future plantation development; in addition, authorities seem to have made a decision to manage large areas of natural forest which has limited the growth of the industry over the past decade.

Finally, expansion of the required number of hectares into low biomass land cover types (shrub, grassland, and agroforestry systems) in Sabah and Sarawak was not feasible under either the MRT or RET scenarios because of a lack of available land in these land cover types. Therefore, we forced the model to replace almost all agricultural land in the case of Sabah and shift expansion into upland forest in Sarawak, in order to avoid developing peat soil areas in the MRT and RET scenarios. Stopping development on high biomass landscapes in Sabah, for example, while maintaining the rates of expansion specified by scenario assumptions caused oil palm to replace other crops and foodstuffs grown on agricultural land, which might shift this production elsewhere and lead to deforestation via indirect LUC.

Performance

Oil palm concessions are very specific to the development of oil palm, and therefore are a very good predictor of expansion. Different factors can be weighted in GEOMOD to represent their different level of importance. Oil palm concessions[3], then palm oil mills, were the two best predictors; unfortunately data on palm oil mill locations were available only for Kalimantan and Sumatra, and concession data were available only for Kalimantan, Sumatra, Papua and Sarawak. Palm oil mills are also backward looking, as they are an indicator of past development, while oil palm concessions are forward looking, since they may indicate the location of future palm oil mills and oil palm plantations. These are important factors that impact the accuracy of the model and including these data layers for all regions would improve model accuracy. Although concessions were an important factor for predicting oil palm expansion, not all expansion occurs within concessions (Appendix 1); therefore, we did not constrain the model to expand only into concession areas, nor do we assume that all landscapes within oil palm concessions will be developed as oil palm plantations. (Both of these were key assumptions of Carlson *et al.,* 2012.) Instead, we based oil palm expansion on historic rates and assumed they would remain on the same trajectory for several years and then decline as global population and consumption patterns stabilized.

CONCLUSIONS

In this study we demonstrate that continuing current policies and practices for oil palm development in Southeast Asia will lead to the continued conversion of high biomass landscapes and peat soils. If future development follows the same business logic as it has historically, then emissions will continue to grow in a linear fashion, resulting in large and climatically significant carbon emissions. These emissions can be reduced substantially by stopping the expansion of oil palm onto peat soils and avoiding high biomass landscapes. We also demonstrate that the drainage of swamp habitats and the oxidation of peat will lead to very large future emissions linked to the palm oil sector, and the equally substantial role that restoration of these areas can play in climate change mitigation.

We used the GEOMOD model to predict locations of oil palm expansion for the purpose of evaluating the potential carbon impacts of different policy options. We also created threat maps that show locations of high risk that can serve to inform policymakers about areas on peat and/or in high biomass areas with a high probability of conversion to oil palm. If the palm oil sector wishes to lower greenhouse gas (GHG) emissions, it should seek to expand into low biomass areas on mineral soils. The projected BAU expansion to 2050 exceeds the available supply of sustainable area in many regions. Alternate scenarios that halt development on peat in the MRT scenario and which displace plantations from peat in the RET scenario force expansion into almost all the remaining 'sustainable' areas by 2050.

We demonstrate that it is possible to shift oil palm expansion to non-peat and low biomass areas that reduce emissions substantially compared to the BAU scenario, and we provide maps of where this expansion could – and perhaps more importantly should not – occur. Not all scenarios modeled in this analysis are realistic to implement, however, and this became apparent after we began the simulation process. In some instances, the forced expansion of oil palm plantations onto low biomass landscapes displaced other forms of agriculture. If such a policy could be developed and implemented, it would have important implications for food security and may lead to indirect LUC, particularly if it displaced rural communities that occupied these previously deforested, but under productive, land cover types (Koh & Ghazoul, 2010).

We conclude that oil palm expansion in Southeast Asia could proceed with a lower emissions profile. Policies that motivate producers to shift to low biomass landscapes on mineral soils and to end all development on peat are shown to be feasible options within the growth projections of the industry. Further reductions in the GHG footprint of the sector can be achieved by retiring existing plantations on peat forest at the end of their current 25-year planting cycle, which would transform the industry and reduce its impact on the atmosphere without sacrificing levels of production.

[3] These data were obtained from numerous sources but no verification of the data's accuracy has been carried out.

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LAND USE CHANGE IN MALAYSIA

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INTRODUCTION

Geographically Malaysia is divided into two regions, namely Peninsular Malaysia and East Malaysia. Peninsular Malaysia is located south of Thailand, north of Singapore and east of the Indonesian island of Sumatra. East Malaysia is located on the island of Borneo and shares borders with Brunei and Indonesia. There are 11 states and two Federal Territories in Peninsular Malaysia, and two, Sabah and Sarawak, in East Malaysia. The total land area of the country is about 329,847 km² Peninsular Malaysia makes up 132,090 km² or 39.7%, while East Malaysia covers 198,847 km², or 60.3% of the total land of the country. It is the only country to contain land on both mainland Asia and the Malay Archipelago.

Malaysian forests can be categorized as tropical rainforests and comprise a variety of types, including dry inland dipterocarp, peat swamps, and mangrove forests. The majority of the forests (about 93%) are upland rainforests dominated by dipterocarp species 5% are peat swamp, and 2% are mangroves. Depending on the altitude, soil conditions and water regime, dipterocarp forest is found in interior upriver areas up to an altitude of 1,500 m, peat swamp forests are found in the low-lying coastal plains, and mangrove forest is found in tidal and estuarine stretches extending from mud flats up to where the saline waters end.

There is also plantation forest established in the country to overcome possible shortage of future timber supply from natural forest. Malaysia is committed to managing forests in a sustainable manner, not just for economic reasons but more importantly for maintaining environmental stability, ecological balance and meeting social obligations. Forest area in Malaysia stands at 19.52 Mha (59.5 % of total land area), with 5.88 Mha in Peninsular Malaysia, 4.40 Mha in Sabah, and 9.24 Mha in Sarawak. Much of the forests are gazetted as permanent forest reserves, national and state parks or other legally protected forests in an effort to conserve the various types of forests for future generations (Table 1). Table 2 shows the total forested area in Peninsular Malaysia from 1970 to 2009. In 2009 alone, the area covered by forest in Peninsular Malaysia amounted about 44.7 percent of the total land area.

Table 1. Land and forest areas in Malaysia in 2008 (million ha)

Region	Land area	Natural Forest			Plantation forest	Total Forested land	% Forest
		Dry inland	Swamp forest	Mangrove forest			
Peninsular Malaysia	13.16	5.40	0.30	0.10	0.08	5.88	44.7
Sabah	7.37	3.83	0.12	0.34	0.11	4.40	59.7
Sarawak	12.30	7.92	1.12	0.14	0.06	9.24	75.1
Total	**32.83**	**17.15**	**1.54**	**0.58**	**0.25**	**19.52**	**59.5**

Source: FRA (2010)

Table 2. Forested areas in Peninsular Malaysia from 1970 – 2009.

Year	Forested area (ha)				
	Permanent reserve forest	Stateland	Wildlife reserve	Other reserve area	Total
1970	3,337,900	4,018,700	656,000	-	8,012,600
1971	3,307,900	3,896,200	674,100	-	7,878,200
1972	3,437,400	3,518,200	634,100	-	7,589,700
1973	3,358,100	3,348,900	634,100	-	7,341,100
1974	3,411,400	3,295,400	611,300	-	7,318,100
1975	3,448,500	3,230,600	611,300	-	7,290,400
1976	2,924,400	3,653,100	611,300	-	7,198,800
1977	3,164,500	3,226,600	576,700	-	6,967,800
1978	2,948,500	3,288,900	602,500	-	6,839,900
1979	2,933,000	2,908,500	603,900	-	6,445,400
1980	3,124,900	2,632,600	603,900	-	6,361,400
1981	3,083,300	2,611,200	599,000	-	6,293,500
1982	3,019,900	2,604,900	556,100	-	6,180,900
1983	4,830,500	1,341,800	555,900	-	6,728,200
1984	5,049,700	682,700	549,800	-	6,282,200
1985	5,103,600	701,400	549,800	-	6,354,800
1986	5,127,300	783,300	549,000	-	6,459,600
1987	4,990,000	813,600	544,200	-	6,347,800
1988	4,928,600	815,000	544,200	-	6,287,800
1989	4,866,201	906,212	548,000	-	6,320,385
1990	4,866,470	853,354	550,173	-	6,269,897
1991	4,748,057	717,277	645,220	-	6,110,554
1992	4,717,732	716,371	607,979	-	6,042,082
1993	4,698,459	717,479	608,070	-	6,024,008
1994	4,687,463	594,433	611,340	-	5,893,236
1995	4,684,904	590,486	611,340	-	5,886,730
1996	4,684,094	521,528	614,925	-	5,820,547
1997	4,731,927	504,025	611,692	5,225	5,852,869
1998	4,730,216	478,409	611,692	18,543	5,838,860
1999	4,853,646	387,090	645,217	52,115	5,938,068
2000	4,837,500	444,817	650,302	47,030	5,979,649
2001	4,840,431	433,674	650,302	-	5,924,407
2002	4,701,858	422,049	768,994	-	5,892,901
2003	4,696,211	413,344	770,168	-	5,879,723
2004	4,683,505	413,664	769,707	-	5,866,876
2005	4,711,264	413,664	763,262	-	5,888,190
2006	4,726,182	413,573	761,634	-	5,901,389
2007	4,695,630	444,991	700,574	-	5,841,195
2008	4,815,529	332,271	586,378	118,026	5,852,204
2009	4,930,569	327,661	615,209	325	5,873,764

Sources:

Forestry Department, Peninsular Malaysia (1971-1978)

Forestry Department, Peninsular Malaysia (1993)

Forestry Department, Peninsular Malaysia (1995)

Forestry Department, Peninsular

Forest Types

The classification of forests in the three regions in Malaysia is very similar. In general, the vegetation changes with altitude from coastal beach forest and mangrove to lowland dipterocarp forest, hill dipterocarp forest and eventually montane forest. The major forest types in the three regions are as follows:

Peninsular Malaysia

- Montane ericaceous forest (>1500m a.s.l.)
- Montane oak forest (1200-1500m a.s.l.)
- Upper dipterocarp forest (750-1200m a.s.l.)
- Hill dipterocarp forest (300-750m a.s.l.)
- Lowland dipterocarp forest (0-350m a.s.l)
- Peat and freshwater swamp forest
- Marine (mangrove) swamp forest

Sabah

- Beach forest.
- Mangrove Nipah forest
- Swamp forest
- Dipterocarp forest (up to 900m a.s.l.)
- Riverine forest.
- Heath forest or 'Kerangas'
- Cloud forest (2,200m a.s.l.)
- Ultramafic forest
- Montane forest (≥ 900m a.s.l.)
- Sub-alpine forest (occurs only on Mt Kinabalu above 3,300 m).

Sarawak

- Hill mixed dipterocarp forest
- Peat swamp forest: -five peat swamp forest types are recognised:
  - Mixed swamp forest
  - Alan Batu forest
  - Alan Bunga forest
  - Padang Batu forest
  - Padang Paya forest
- Mangrove forest
- Kerangas
- Montane Forest

Biological diversity

Ecologically, Malaysia is biologically rich nation with a diverse range of flora and fauna found in various eco-regions throughout the country. It is home to 15500 species of higher plants, 746 birds, 379 reptiles, 198 amphibians, and 368 species of fish. There are also 286 species of mammals in Malaysia, of which 27 are endemic and 51 are threatened. Some of these mammals are found in both Peninsular Malaysia and Malaysian Borneo. The former has 193 species of mammals, while the latter has 215. Among the mammals that are native to Malaysia include the Asian elephant, the Indochinese tiger, and the Leopard cat. Endangered species include the orangutan, the tiger, the Asian elephant, the Malayan tapir and the Sumatran rhinoceros. The tropical forests of Peninsular Malaysia contain 450 species of birds and over 6000 different species of trees, of which 1000 are vascular plants.

Oil Palm in Malaysia

Malaysia is a developing country and the agriculture sector is still considered to be a backbone for income generation. Large areas of land are currently planted with agriculture commodities such as oil palm, rubber and rice. Malaysia is one of the largest exporters of palm oil in the world producing 15.8 Mt of crude palm oil in 2007. It is also one of the largest producers and exporters of rubber and other natural rubber products.

Oil palm was first introduced into Malaysia in 1917, as a prime crop choice for the agriculture diversification program. In Peninsular Malaysia, oil palm areas expanded from a mere 55,115 ha in 1970 to 906,590 ha in 1980, over a span of only one decade (Table 3). The oil palm planted area later doubled to 1.7 Mha by 1990. Currently, the cultivation of oil palm in Peninsular Malaysia covers an area of about 2.5 Mha and the total area planted with oil palm for the whole of Malaysia in 2009 was about 4.7 Mha.

Table 3. Oil palm plantation area (ha) in Malaysia 1975 – 2009

Year	Peninsular Malaysia	Sabah	Sarawak	Total
1975	568,561	59,139	14,091	641,791
1976	629,558	69,708	15,334	714,600
1977	691,706	73,303	16,805	781,814
1978	755,525	78,212	19,242	852,979
1979	830,536	86,683	21,644	938,863
1980	906,590	93,967	22,749	1,023,306
1981	983,148	100,611	24,104	1,107,863
1982	1,048,015	110,717	24,065	1,182,797
1983	1,099,694	128,248	25,098	1,253,040
1984	1,143,522	160,507	26,237	1,330,266
1985	1,292,399	161,500	28,500	1,482,399
1986	1,410,923	162,645	25,743	1,599,311
1987	1,460,502	182,612	29,761	1,672,875
1988	1,556,540	213,124	36,259	1,805,923
1989	1,644,309	252,954	49,296	1,946,559
1990	1,698,498	276,171	54,795	2,029,464
1991	1,744,615	289,054	60,359	2,094,028
1992	1,775,633	344,885	77,142	2,197,660
1993	1,831,776	387,122	87,027	2,305,925
1994	1,857,626	452,485	101,888	2,411,999
1995	1,903,171	518,133	118,783	2,540,087
1996	1,926,378	626,008	139,900	2,692,286
1997	1,959,377	758,587	175,125	2,893,089
1998	1,987,190	842,496	248,430	3,078,116
1999	2,051,595	941,322	320,476	3,313,393
2000	2,045,500	1,000,777	330,387	3,376,664
2001	2,096,856	1,027,328	374,828	3,499,012
2002	2,187,010	1,068,973	414,260	3,670,243
2003	2,202,166	1,135,100	464,774	3,802,040
2004	2,201,606	1,165,412	508,309	3,875,327
2005	2,298,608	1,209,368	543,398	4,051,374
2006	2,334,247	1,239,497	591,471	4,165,215
2007	2,362,057	1,278,244	664,612	4,304,913
2008	2,410,019	1,333,566	744,372	4,487,957
2009	2,489,814	1,361,598	839,748	4,691,160

Source: Statistics on Commodities, Malaysian Oil Palm Statistics (2010)

MATERIALS AND METHODS

Table 4: Summary of data sources.

Region	Year/soil type	Data Source	Remarks
Peninsular Malaysia	1990	DOA[a]	Land use map in GIS format. The land use categories were re-grouped into 6 classes focusing on the major land use types including oil palm plantations.
	2006	DOA[a]	Land use map in GIS format. The land use categories were re-grouped into 6 classes focusing on the major land use types including oil palm plantations.
	2009	MPOB[b]	Oil palm distribution based on satellite images produced by MPOB. Since data provided by MPOB only cover the extent and distribution of oil palm areas, analysis of land use change (LUC) was limited to oil palm areas.
	Peat	DOA[a]	
Sarawak	2000	MACRES[c]	The source is a land cover map prepared by MACRES using satellite images. The map was digitized and land uses were re-grouped into 6 classes including oil palm plantations. This is the base map used to compare with the other recent data
	2005	FRIM[d]	The source consisted of Landsat TM images covering the whole of Sarawak. Land cover classifications were made and updated based on the base map of 2000. Digitizing and analyzing of the latest (2005) land covers were made on the 2005 images.
	2009	MPOB[b]	Oil palm distribution data were based on satellite images produced by MPOB. Since data provided by MPOB only included the extent and distribution of oil palm areas, analysis of LUC was limited to these areas.
	Peat	DOA[a]	
Sabah	2000	MACRES[c]	The source is a land cover map prepared by MACRES using satellite images. The map was digitized and land uses were re-grouped into 6 classes including oil palm plantations. This is the base map used to compare with the other recent data.
	2005	FRIM[d]	The source consisted of Landsat TM images covering the whole of Sabah Land cover classifications were made and updated based on the base map of 2000. Digitizing and analyzing of the latest (2005) land covers were made in the 2005 images.
	2009	MPOB	Oil palm distribution data were based on satellite images produced by MPOB. Since data provided by MPOB only included the extent and distribution of oil palm areas, analysis of LUC was limited to these areas.
	Peat	DOA[a]	

[a] *DOA (Department of Agriculture, Malaysia)*
[b] *MPOB (Malaysian Palm Oil Board)*
[c] *MACRES (Malayan Centre for Remote Sensing)*
[d] *FRIM (Forest Research Institute of Malaysia)*

The main aim of the study was to make an assessment of land use change trends in Malaysia by comparing various geo-spatial data (including satellite images and other relevant maps). The image analysis was done in laboratory and no field verification was carried out. Therefore, the area extent figures may be different from the actual areas on the ground.

Data sources

The study is divided into three main regions, namely Peninsular Malaysia, Sarawak and Sabah. The data sources used in the study also vary according to the regions as listed in Table 4.

Study Area

Based on its geographical regions, the study in Malaysia is divided into three parts:

i) Peninsular Malaysia
ii) Sarawak
iii) Sabah

Peninsular Malaysia

Three main data sources were used for the land use analysis of Peninsular Malaysia as indicated in Table 4. The land use categories in each of the 1990 and 2006 data sets were re-grouped into six major land use categories. This was done mainly on the assumption that only these land use categories have significant influence on the expansion of oil palm areas in the country. Figure 1 shows the general methodology used in the land use change matrix study for Peninsular Malaysia. Due to time and budget constraints, no ground survey was undertaken. However, high resolution images available from Google Earth were used wherever possible to verify the choice of land use category.

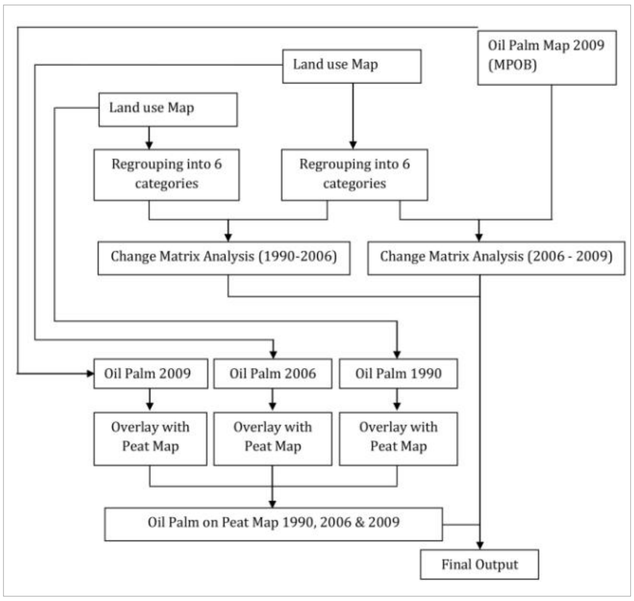

Figure 1. General flow of land use change matrix analysis for Peninsular Malaysia

Sabah and Sarawak

For Sabah and Sarawak the data sets used are presented in Table 4. The land use maps were mainly prepared using Landsat images. The 2000 map was prepared based on the Satellite Image Atlas prepared by MACRES as a hard copy. The map was converted to digital form for the digital land use change analysis. For Sabah and Sarawak this 2000 map was used as the base map for the study. Figure 2-shows the general approach used in land use change matrix analysis for Sabah and Sarawak.

In order to fulfil the project objectives, 15 scenes of Landsat images of Sabah and Sarawak were processed using the methods described in the following sections. The activities involved in this study were data acquisition, data pre-processing, data processing, and accuracy assessment.

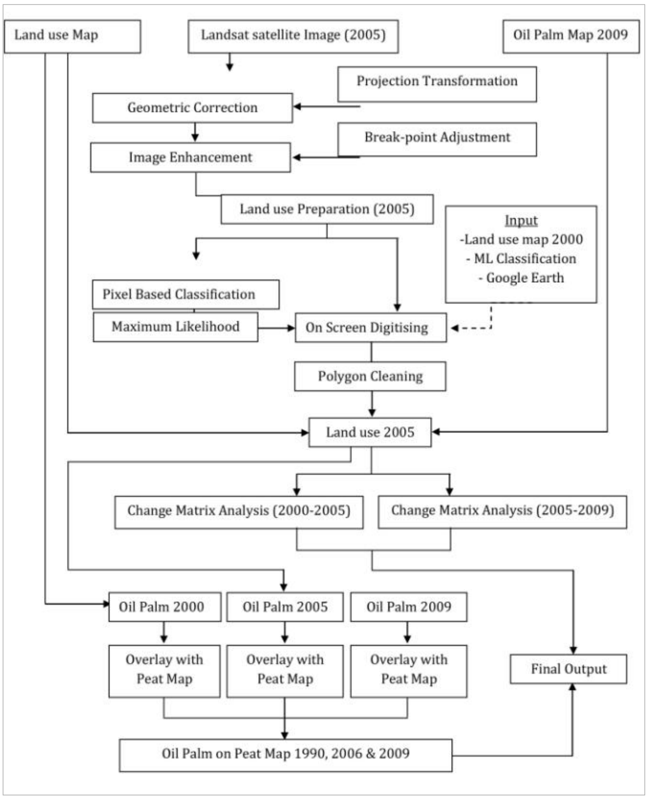

Figure 2. General flow of Land use change matrix analysis for Sabah and Sarawak

Methodology

Data Acquisition

The primary datasets used consisted of multispectral Landsat images used to classify the main land uses in Sabah and Sarawak. Landsat sensors collect image data of 30 meter nominal pixel size. The Landsat images are suitable for land use map classification because these data have enough spectral and spatial resolution to discriminate broad land use types. In satellite image land use classification, ground truth data is important for identifying the various land use classes existing on the ground. For this project, the land use classes were identified using information collected from the existing land use map of 2000 and Google Earth applications.

Data Processing

Image processing involves geometric correction, image enhancement, and clipping (sub-setting) images to give areas of interest.

a) Geometric correction

Geometric correction is applied to all raw datasets to correct errors of perspective due to the Earth's curvature and sensor motion. In this project, the Landsat images were geometrically corrected using the 'Projection Transformation" tool found in ERDAS Imagine software. The images were projected to give Rectified Skewed Orthomorphic (RSO) views with datum from the Modified Everest map projection system. The images were then resampled using the Nearest Neighbour scheme because of its abilty to preserve the original digital numbers.

b) Image enhancement

Image enhancement is a process of improving feature interpretability through various techniques, such as adjusting brightness and contrast. The values used to adjust brightness and contrast were stored in a breakpoint look up table (LUT). LUTs should be created to optimize interpretability of features of interest. Enhancements should also involve optimizing gray level balance of panchromatic bands and colour balancing of multispectral bands between adjacent scenes, while maintaining the variability in the source.

c) Data Classification

The most important process in this study was the classification of the satellite images to identify land use classes. Supervised maximum likelihood classification technique was performed. Seven land use classes were defined (Table 5) which could be separated using moderate spatial resolution satellite imagery. The major component of supervised classification is the creation of training areas, which are controlled by the user. In this process, identified pixels that represent patterns or land use classes were selected with the help of Google Earth data and information from old land use maps. In this method, knowledge of the data and of the classes desired is required before classification proceeds. By identifying patterns, the computer will identify pixels with similar characteristics. The Maximum Likelihood decision rule is based on the probability that a pixel belongs to a particular class. The basic equation assumes that these probabilities are equal for all classes and that the input bands have normal distributions. If it is known that the probabilities are not equal for all classes, a weight factor for particular classes can be specified. This variation of the maximum likelihood decision rule is known as the Bayesian decision rule. Unless a priori knowledge of the probabilities is available, it is recommended that they not be specified. In this case, these weights default to 1.0.

Table 5. Land use classes based on supervised classification

Land use Class	Land use Name
1	Built up area/Urban
2	Rubber
3	Oil palm
4	Wetland
5	Horticulture
6	Forest
7	Unclassified (Cloud, Shadow)

d) Vector Data Editing

Polygon cleaning is needed to construct topology for the vector layer. The cleaning process includes both lines and polygons. This is a slow process, but should be used if the layer has undergone major edits. In this process, if the input file contains attribute information, it is automatically updated. If the layer does not contain attribute information, this is created.

e) Change Matrix Analysis

After polygon cleaning is done, the vector data are split into their classes. The splitting process separates the merge classification vector into individual themes so that the analysis of land use data can be made easier. Every class is inserted with its own attribute which can be used to determine the class name, type, area, and individual code. Change matrix analysis was done for both the period between 2000 (using the old land use map) and the final output of digitised in 2005 and also for changes of land use by oil palm from 2005 to 2009.

Oil Palm Area on Peat

All of the oil palm data (2000, 2005, and 2009) in vector format was overlaid with the peat distributions maps of Sabah and Sarawak. This process was done to calculate the extent of oil palm on peat.

RESULTS AND DISCUSSION

Peninsular Malaysia

Major land uses

The extent of major land uses in Peninsular Malaysia is given in Table 6 which shows that the total oil palm area increased from 1.4 Mha in 1990 to about 2.58 Mha in 2009. Forest and rubber areas decreased from 6.52 Mha to 6.0 Mha and from 2.28 Mha to 1.54 Mha respectively between 1990 and 2006. There was no information for those categories in 2009. Figure 3 shows the respective land use maps for the year 1990 and 2006, as well as the oil palm distribution in Peninsular Malaysia in 2009.

Table 6. The extent of land use in Peninsular Malaysia in 1990, 2006 and 2009.

Land use Category	Area in 1990 (ha)	Area in 2006 (ha)	Area in 2009 (ha)
Oil Palm	1,418,263	2,545,893	2,683,217
Rubber	2,279,001	1,535,127	Na
Built up area	385,186	530,931	Na
Wetland	954,643	562,128	Na
Forest	6,522,499	6,007,838	Na
Others	1,573,844	1,994,369	Na

Na = not available
Built up includes farm, buildings and associated areas, urban, residential, and tin mining areas
Forest includes all forest types including reserves and state land forests.
Wetland includes all swamps (peat swamp, fresh water swamp, mangrove, etc).
Others include cocoa, orchards, unused land, mixed horticulture, aquaculture, coffee, black pepper, pineapples, other crops, sago palm, sugarcane, tea, and paddy.

Land Use Change Matrix

A land use change matrix analysis for different years in Peninsular Malaysia was carried out in order to determine the land use change trend. This can be used to identify which land use categories contributed to the increasing area of oil palm in the country within a given time frame. Table 7 shows the land use change matrix for the years 1990 and 2006 for Peninsular Malaysia. It shows that about 23% (587,792 ha) of the newly planted oil palm land present in 2006 originally came from rubber, whereas forest only contributed about 14% (364,457 ha). Since conversion of forested areas can only be carried-out in state land forests, it is assumed that minimum Permanent Forest Reserve (PFR) conversion took place during this time. Based on this analysis, most of the oil palm conversion areas were from rubber. Due to the data limitations, a detailed land use change matrix for the period 2006 to 2009 could not be carried out. However, the trend in the oil palm area during this period is indicated in Table 8. The results show that about 4% (120,068 ha) of the newly planted oil palm area found in 2009 came from the forest land use category, while in contrast, about 9.5% (254,433 ha) of the oil palm area in 2009 was converted from rubber.

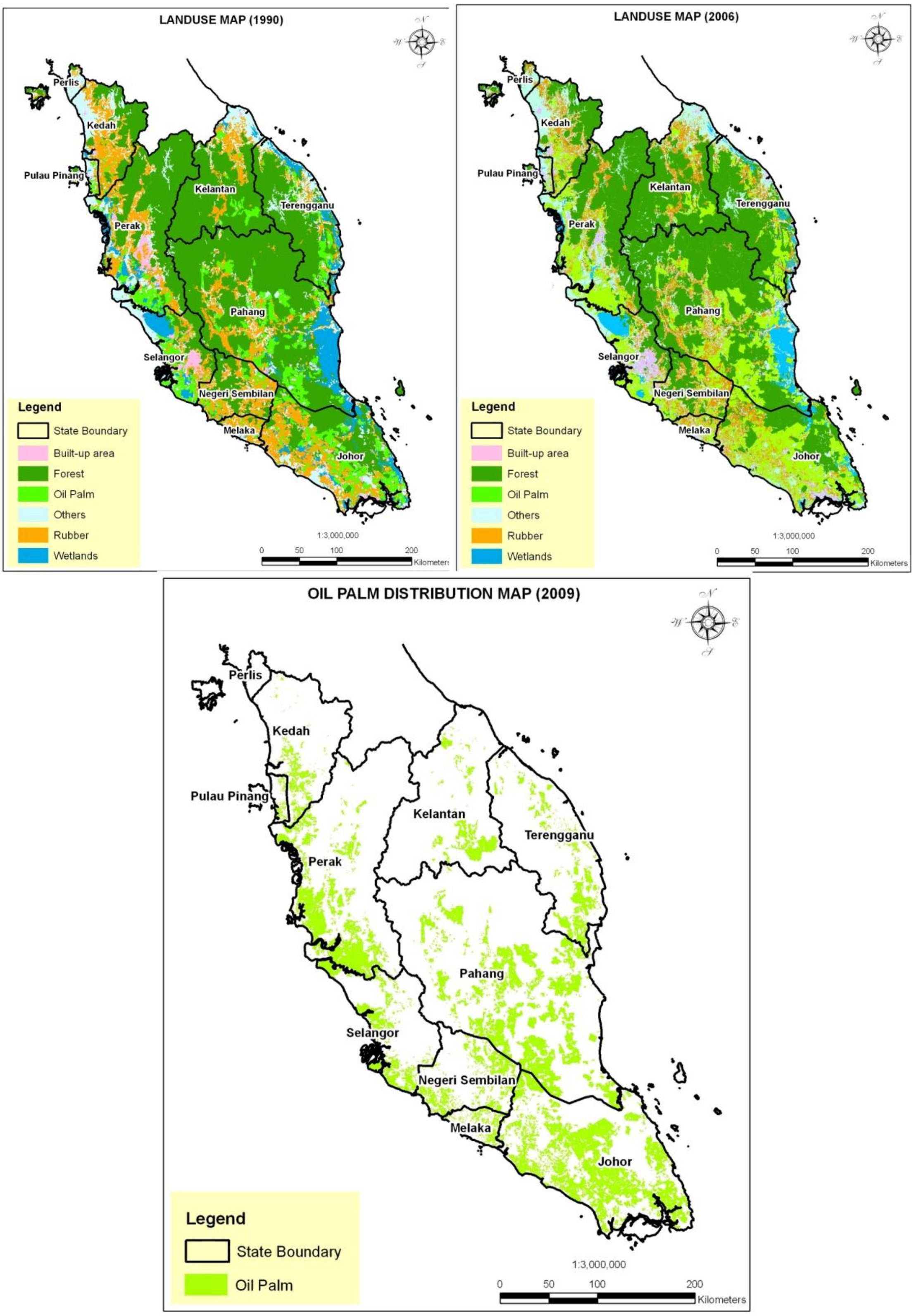

Figure 3. Land use in Peninsular Malaysia in 1990 (top left), 2006 (top right), and the distribution of oil palm plantations in 2009 (bottom).

Table 7. Land use change matrix for the period 1990 to 2006 in Peninsular Malaysia.

Land use categories	Land use change (ha)						Land use in 2006 (ha)
	Built-up area	Forest	Oil Palm	Others	Rubber	Wetlands	
Oil Palm	53,094	49,120	1,158,873	89,743	53,670	13,576	**1,418,076**
Rubber	150,885	95,397	587,792	397,179	1,036,187	9,713	**2,277,154**
Built-up area	140,673	22,226	72,898	113,041	26,853	9,641	**385,332**
Wetland	28,605	101,757	162,811	145,699	28,977	478,320	**946,168**
Forest	46,359	5,577,840	364,457	301,857	200,378	20,404	**6,511,296**
Others	108,326	156,154	198,605	890,628	187,709	21,305	**1,562,727**
Land use in 1990 (ha)	**527,941**	**6,002,494**	**2,545,437**	**1,938,146**	**1,533,774**	**552,959**	**13,100,752**

Table 8. Land use change matrix for Peninsular Malaysia for the years 2006 to 2009.

Land use categories	Oil Palm area (ha)				Oil Palm in 2009 (ha)
	Immature	Mature	Old	Unclassified	
Oil Palm	385,349	1,595,664	59,582	504,842	**2,545,437**
Rubber	57,233	190,192	7,009	Na	**254,434**
Built up area	3,816	20,047	707	Na	**24,569**
Wetland	16,289	17,522	1,098	Na	**34,909**
Forest	65,732	51,699	2,638	Na	**120,068**
Others	34,092	167,371	6,855	Na	**208,318**
Land use in 2006 (ha)	**562,510**	**2,042,495**	**77,888**	**504,842**	**2,682,894**

Oil palm on Peat

The area of peat in Peninsular Malaysia is about 717,347 ha. By using the peat distribution map and land use maps of Peninsular Malaysia, an estimate of the area of oil palm planted on peat was obtained. Table 9 shows the areas of oil palm planted on peat in different years in Peninsular Malaysia. It shows that less than one percent of peat was developed for oil palm plantations in 1990. However, this increased to 8.3% in 2006 followed by an almost insignificant increase of about 0.1% by 2009. This shows that conversion of peat forest for planting oil palm in Peninsular Malaysia is no longer a serious problem. Figure 4 shows the distribution of oil palm planted on peat in Peninsular Malaysia for the different years.

Table 9. The area of oil palm planted on peat in Peninsular Malaysia in 1990, 2006 and 2009.

Year	Oil palm area on peat (ha)	Total oil palm area (ha)	% oil palm on peat
2000	111,954	1,418,263	7.9
2005	212,925	2,545,893	8.3
2009	226,533	2,683,217	8.4

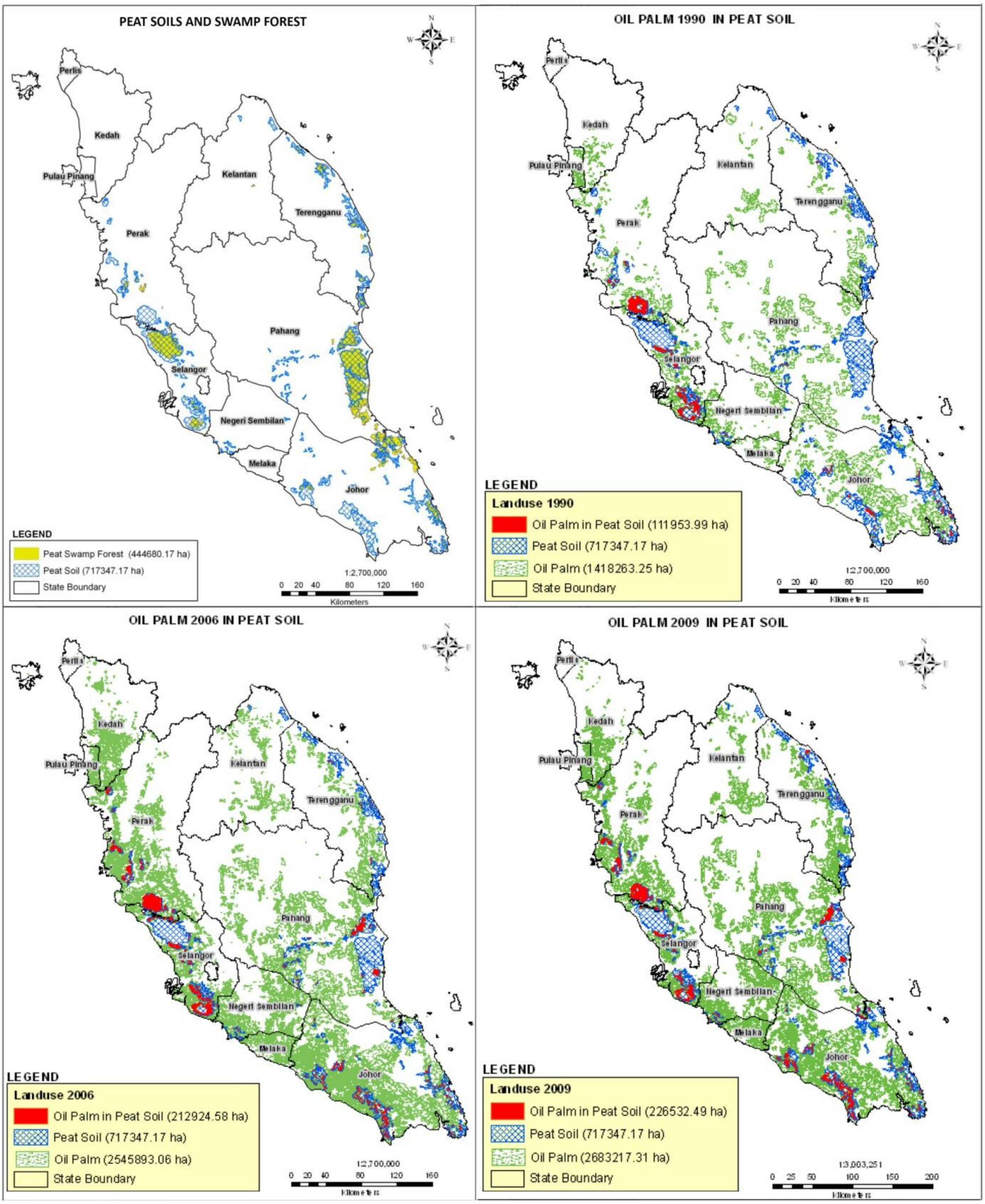

Figure 4. The distribution of peat soils and remnant swamp forests in Peninsular Malaysia (top right); oil palm planted on peat soil 1990 (top left); 2006 (bottom left) and 2009 (bottom right).

Sarawak

Major land use

The extent of major land use in Sarawak is given in Table 10. Only five main land use categories were considered in this study. The category "Others" refers to land use related to horticulture, shifting cultivation, grassland, bare land, coconut, paddy and features that can't be identified from satellite images alone. It shows that the total oil palm area in Sarawak increased from 473,134 ha in 2000 to about 1.16 Mha in 2009. Figure 5 shows the respective land use maps for the years 2000 and 2005, as well as the oil palm distribution in Sarawak in 2009.

Table 10. Land use in Sarawak in 2000, 2005 and 2009.

Land use Category	Area in 2000 (ha)	Area in 2005 (ha)	Area in 2009 (ha)
Oil Palm	473,134	543,515	1,164,386
Rubber	152,717	209,918	Na
Built up area	11,995	25,389	Na
Wetland	1,343,063	1,251,061	Na
Forest	7,140,871	6,621,643	Na
Others	3,081,011	3,551,264	Na

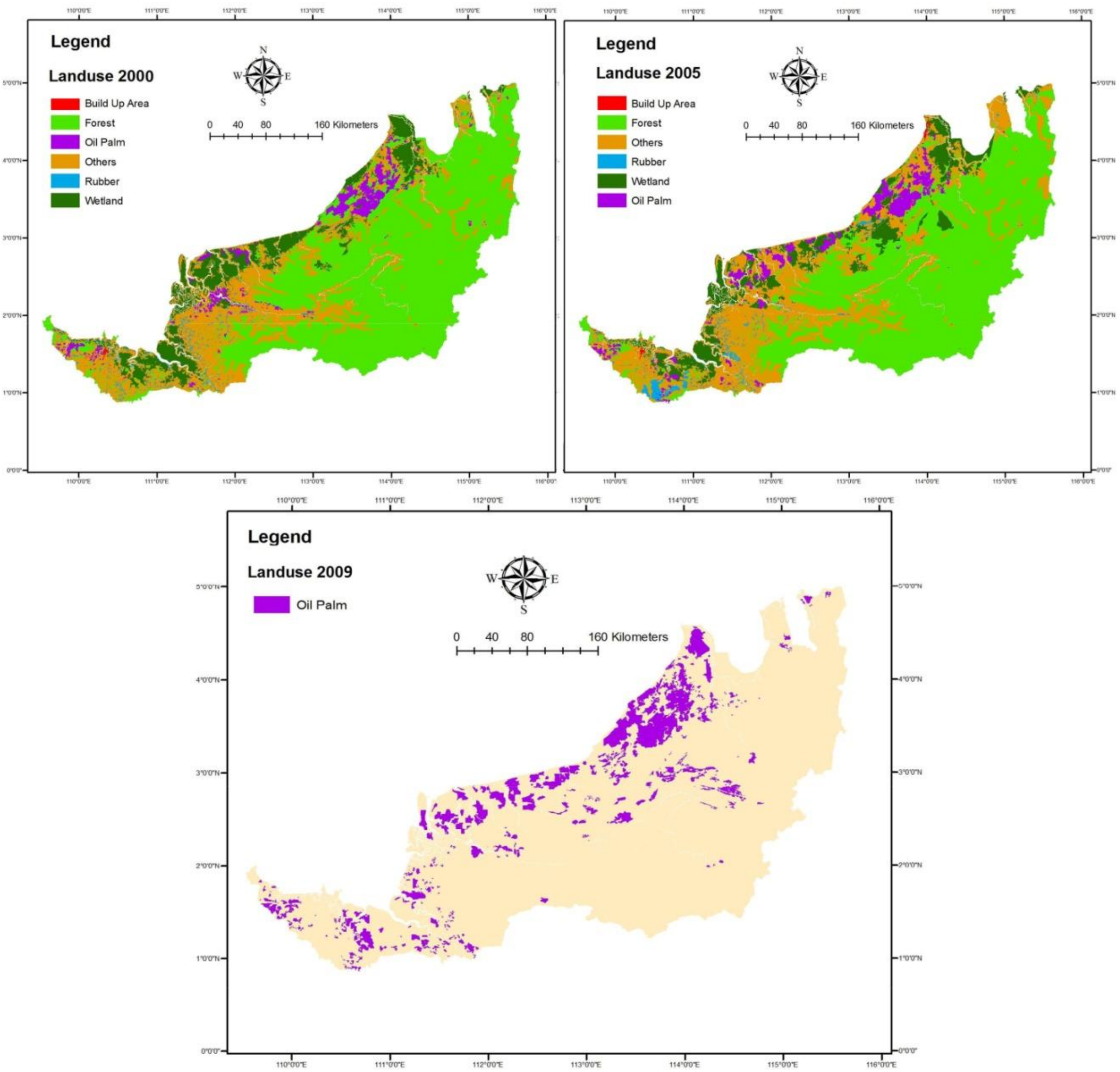

Figure 5. Land use in Sarawak in 2000 (top left), 2005 (top right), and the distribution of oil palm plantations in 2009 (bottom)

Table 11. Land use change matrix for the years 2000 and 2005 for Sarawak.

Land use categories	Land use change (ha)						Land use in 2000 (ha)
	Oil Palm	Rubber	Built up area	Wetland	Forest	Others	
Oil Palm	266,382	1,185	1,228	1,706	3,992	198,641	**473,134**
Rubber	2,040	62,903	164	2,833	521	84,256	**152,717**
Built up area	74	401	10,457	222	271	570	**11,995**
Wetland	186,876	11,383	2,643	854,789	16,330	271,042	**1,343,063**
Forest	46,406	24,820	2,631	213,388	6,500,591	353,035	**7,140,871**
Others	41,738	109,225	8,266	178,123	99,938	2,643,720	**3,081,011**
Land use in 2005 (ha)	**543,515**	**209,918**	**25,389**	**1,251,061**	**6,621,643**	**3,551,264**	**12,202,790**

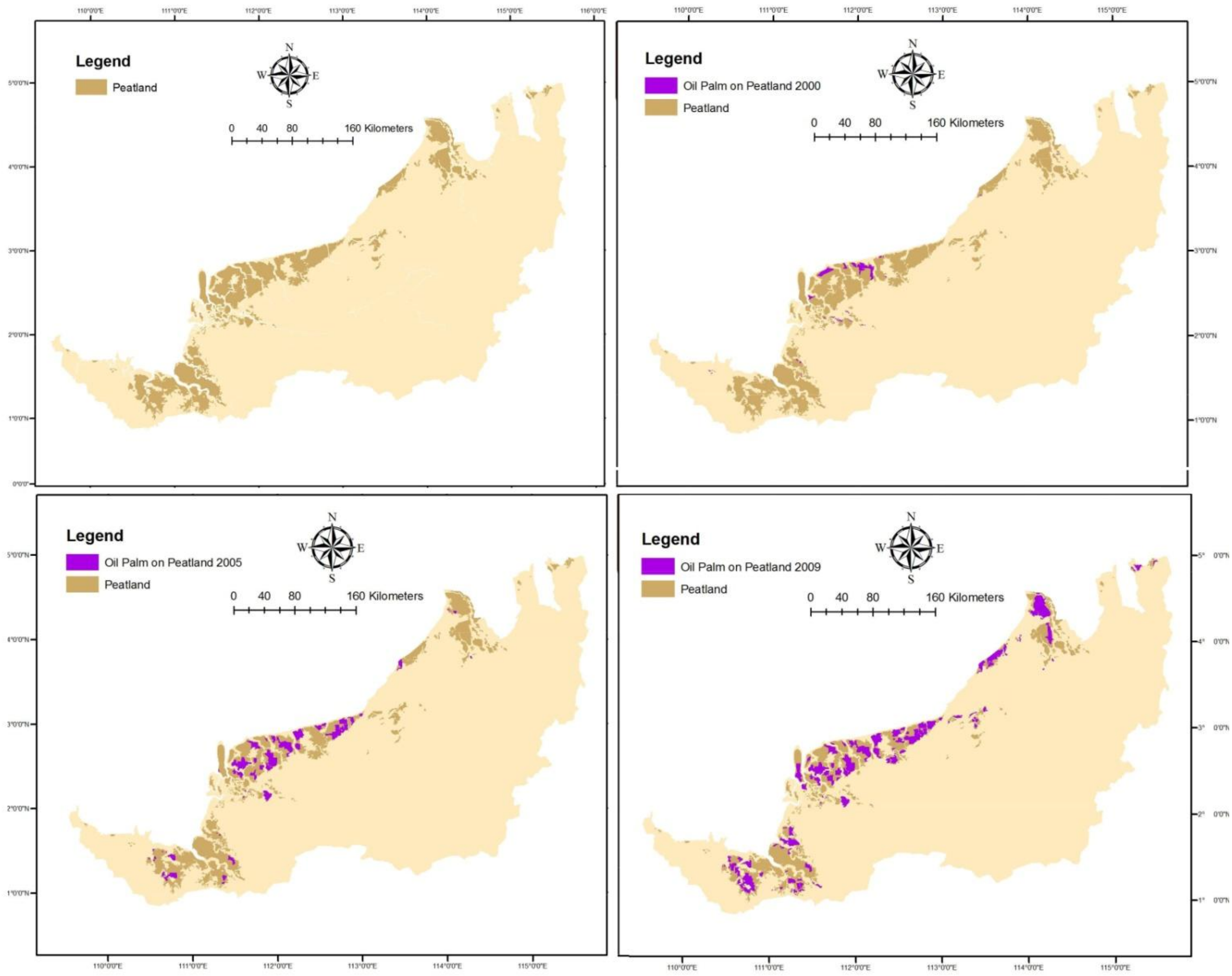

Figure 6. The distribution of peat soils and remnant swamp forests in Sarawak (top right); oil palm planted on peat soil 1990 (top left); 2006 (bottom left) and 2009 (bottom right).

Land use change matrix

A land use change matrix analysis for different years in Sarawak was carried out in order to determine the land use change trend. Table 11 shows the land use change matrix for the years 2000 and 2005. The main land use categories converted to oil palm in this period were wetlands (34% or 186,876 ha). In contrast, the forest land use category only contributed about 8% (46,406 ha).

Oil palm on Peat

The extent of peat in Sarawak is about 1.28 Mha (Figure 6). By using the peatland distribution map and the different land use maps, an estimate of oil palm area established on peat was undertaken. Figure 6 and Table 12 show the area of oil palm planted on peat in different years in Sarawak. It shows an increasing trend from about 3% of the total planted area in 2000 to about 34% in 2009.

Table 12. The area of oil palm planted on peat in Sarawak in 2000, 2005 and 2009.

Year	Oil palm area on peat (ha)	Total area of oil palm (ha)	% oil palm on peat
2000	40,010	473,134	3
2005	193,031	543,515	15
2009	434,057	1,164,386	34

Sabah

Major land use

The extent of major land use in Sabah is given in Table 13. As for Sarawak only five main land use categories were considered in this study. The category under "Others" refers to land use related to horticulture, shifting cultivation, grassland, bare land, coconut, paddy and features that can't be identified from satellite images. It shows that the total oil palm area in Sabah increased from 1,115,020 in 2000 to about 1,452,199 ha in 2009. Figure 7 shows the respective land use maps for the years 2000 and 2005, as well as the oil palm distribution in Sabah in 2009.

Table 13 Land use in Sabah in 2000, 2005 and 2009

Land use Category	Area in 2000 (ha)	Area in 2005 (ha)	Area in 2009 (ha)
Oil Palm	1,115,020	1,151,756	1,452,199
Rubber	78,895	62,891	Na
Built up area	91,003	55,546	Na
Wetland	615,502	435,151	Na
Forest	4,773,501	4,641,200	Na
Others	1,035,580	1,488,620	Na

Land use change matrix

Table 14 shows the land use change matrix for Sabah for the period 2000 to 2005. It shows that the main land use category converted to oil palm was from the "Others" class (8.5% or 97,909 ha) whereas the forest land use category only contributed about 5.5% (63,960 ha).

Table 14 Land use change matrix for Sabah for the period 2000 to 2005.

Land use categories	Land use change (ha)						Land use in 2000 (ha)
	Oil Palm	Rubber	Built up area	Wetland	Forest	Others	
Oil Palm	958,676	1,381	359	621	24,114	129,869	**1,115,020**
Rubber	5,289	48,629	353	26	16,117	8,481	**78,895**
Built up area	535	695	39,588	283	19,507	30,395	**91,003**
Wetland	25,387	391	1,783	417,493	129,590	40,858	**615,502**
Forest	63,960	6,200	4,432	12,334	4,214,350	472,225	**4,773,501**
Others	97,909	5,594	9,032	4,394	237,522	681,128	**1,035,580**
Land use in 2005 (ha)	**1,151,756**	**62,891**	**55,546**	**435,151**	**4,641,200**	**1,362,956**	**7,785,087**

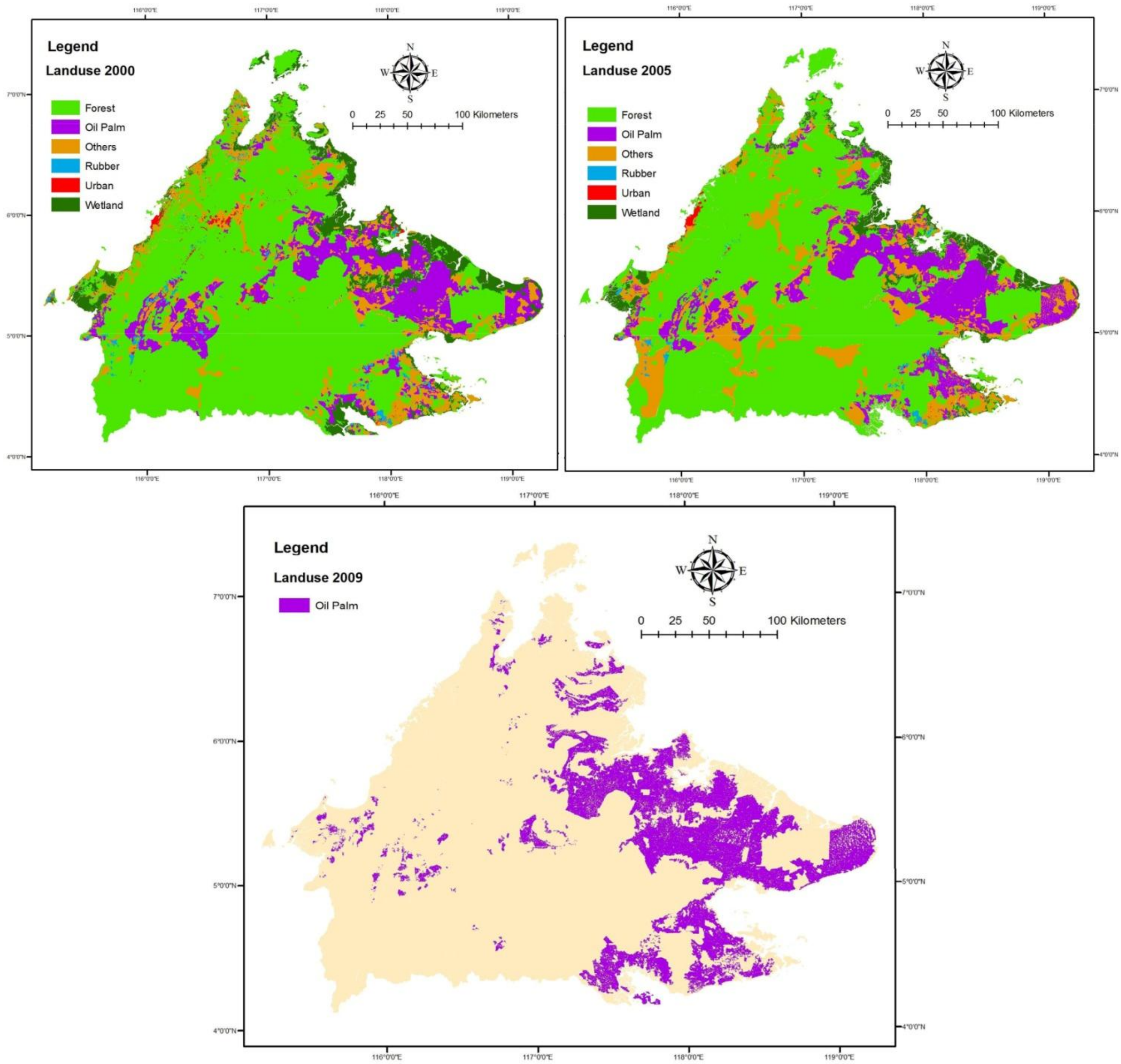

Figure 7. Land use in Sabah in 2000 (top left), 2005 (top right), and the distribution of oil palm plantations in 2009 (bottom).

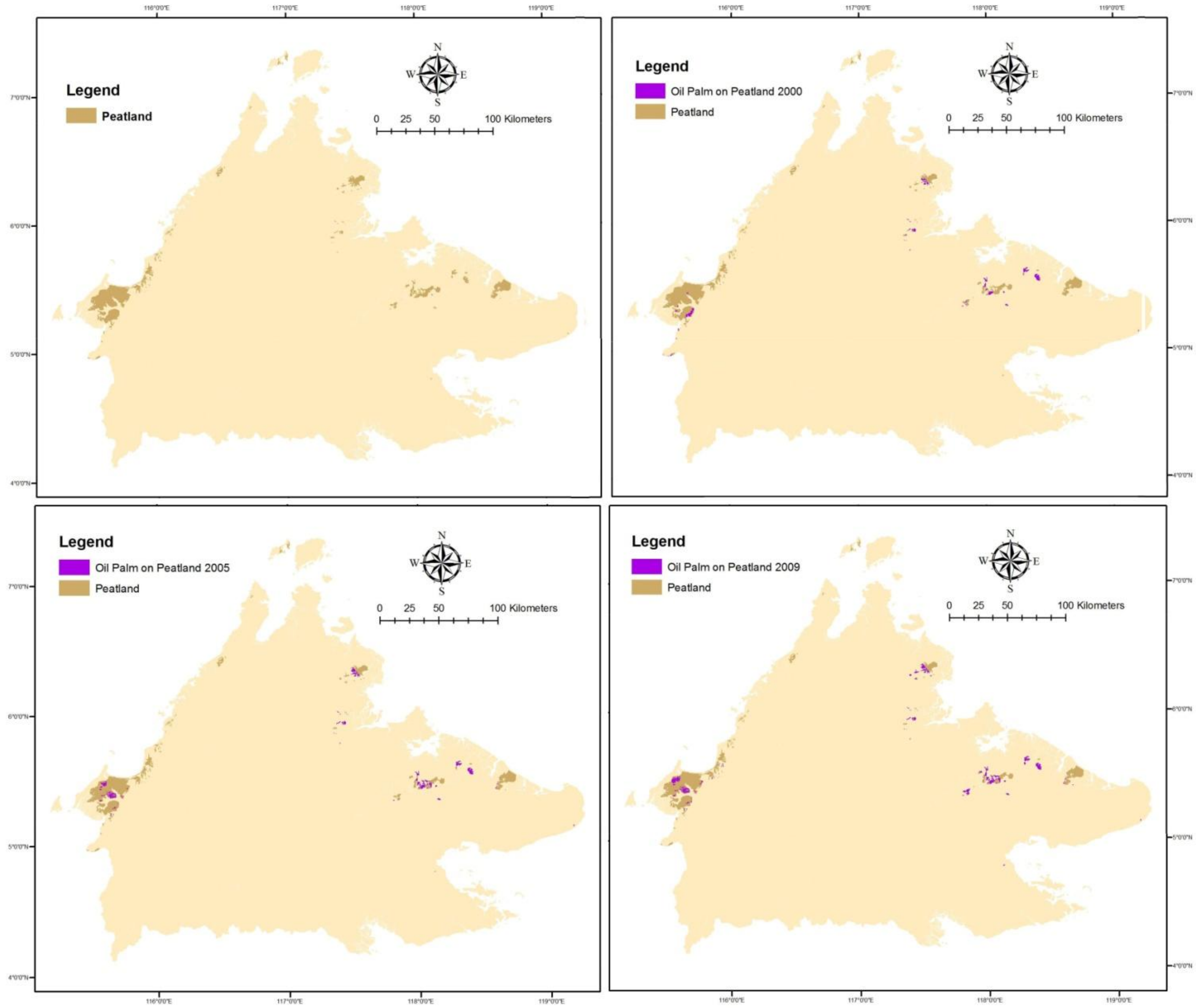

Figure 8. The distribution of peat soils and remnant swamp forests in Sabah (top right); oil palm planted on peat soil 1990 (top left); 2006 (bottom left) and 2009 (bottom right)

Oil palm on Peat

The extent of peat-in Sabah is about 116,965 ha (Figure 8). By using the peat distribution map and the different land use maps, an estimate was made of the area of oil palm on peat (Table 15). This showed an increasing trend over time but was small (<2%) when expressed as a % of the total oil palm area in the State. Figure 8 shows the distribution of oil palm planted on peat in Sabah in the different years.

Table 15. The extent of oil palm planted on peat in Sabah in 2000, 2005 and 2009.

Year	Oil palm area on peat-(ha)	Total oil palm	% oil palm on peat
2000	11,139	1,115,020	1
2005	18,675	1,151,756	1.6
2009	21,043	1,452,199	1.5

CONCLUSIONS

The study shows that from 1990 to 2009 there were substantial changes in land use in Malaysia. Being a developing country this is to be expected and it is anticipated to continue in the near future. In the three regions of the country (Peninsular Malaysia, Sarawak and Sabah), the rate of land use change was found to differ for various reasons including different socio-economic factors in these regions. For Peninsular Malaysia about 23% (587,792 ha) of the newly planted oil palm found in 2006 originally came from ex-rubber land, while forest land contributed only about 14% (364,457 ha). In Sarawak the study shows that the total oil palm area increased from 473,134 ha in 2000 to about 1.16 million hectares in 2009. In this region the main land use category converted to oil palm was from the “wetland” category (34% or 186,875 ha) while the forest land use category only contributed about 8% (46,406 ha). In the case of Sabah the main land use categories converted to oil palm were from the “Others” category (8.5% or 97,909 ha) where as the forest land use category only contributed about 5.5% (63,960 ha).

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ENVIRONMENTAL AND SOCIAL IMPACTS OF OIL PALM CULTIVATION ON TROPICAL PEAT
A SCIENTIFIC REVIEW

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ABSTRACT

This report provides a review of available scientific information and published literature on impacts of using tropical peat for oil palm cultivation in Southeast Asia. It describes carbon flows and greenhouse gas (GHG) emissions from native and degraded forest and oil palm plantations on peat, as well as other environmental impacts and social and economic aspects of the cultivation of oil palm on peat. Based on the available literature, the report presents conclusions on the gaps in knowledge, uncertainties and confusion in existing datasets.

The palm oil sector has created in the past few decades millions of jobs. Over the next decade, the Indonesian government plans to double the annual production of palm oil, creating new jobs for an estimated 1.3 million households. Although the cultivation of oil palm on peatlands creates new income opportunities for many farmers in the short term, longer term economic implications remain uncertain. Transformation of tropical peat forest into plantations will lead to the loss of ecosystem services and biodiversity and will affect the social and cultural basis of forest dependant communities. Human health is affected negatively by haze resulting from forest and peat fires related to land preparation and drainage of the peat. There may be other negative ecological consequences linked to soil subsidence, which can lead to flooding and salt water intrusion when water tables reach levels and the land becomes undrainable.

When peat is developed for agriculture, carbon is lost as CO2 because: 1) oxidation of the peat; 2) fire; and 3) loss from biomass due to land use change. The simplest way to limit CO2 and other GHG emissions is to avoid the development of oil palm plantations on peat. Development of plantations on mineral, low carbon, soils has fewer impacts in terms of GHG emissions. For existing plantations on peat, effective water management (keeping water tables as high as practical) reduces GHG emissions, soil subsidence and fire risk. Nonetheless, even these measures will not turn the system into a carbon or GHG sink.

Keywords: *tropical peat, oil palm cultivation, forests, carbon, greenhouse gases, biodiversity, socio-economic impacts, Southeast Asia.*

INTRODUCTION

Context

On November 4th 2009, a resolution was adopted at the 6th General Assembly of the Roundtable on Sustainable Palm Oil (RSPO) on the 'Establishment of a working group to provide recommendations on how to deal with existing plantations on peat' (Box 1). In the justification for the resolution, it was noted that peat lands are the most efficient and the largest terrestrial carbon store. Accounting for less than 3% of the global land surface, they store more carbon than all terrestrial biomass, and twice as much as all forest biomass. It was mentioned that peat land ecosystems and their natural resources are under great threat as a result of large scale reclamation, deforestation and drainage, causing degradation and the loss of soil carbon by oxidation.

Box 1

Background and objectives of the RSPO Peatland Working Group (PLWG)

The objective of the Roundtable on Sustainable Palm Oil (RSPO) is to promote the growth and use of sustainable palm oil products through credible standards and the engagement of stakeholders. The Peat Land Working Group (PLWG) as part of the RSPO is a short-term multi-stakeholder expert panel established to review the impacts of plantation development and palm oil production in terms of carbon and GHG emissions, as well as any additional effects on biodiversity, livelihoods. The panel seeks to advise the Executive Board regarding actions and processes that will lead to meaningful and verifiable reductions in greenhouse gas emissions in the palm oil supply chain. This review of the scientific literature on the impacts of oil palm plantation development is meant to provide a baseline for recommendations for reducing GHG emissions for palm oil production on peat and its associated management.

The resolution also referred to the first RSPO greenhouse gas (GHG) working group, which had been established to investigate and develop principles and criteria for reducing GHG emissions from land use change, had not been able to reach a consensus on the issue of how to deal with existing oil palm plantations on peat. It was noted that even when assuming minimum estimates of CO_2 emissions from existing oil palm plantations on peat, these plantations were not sustainable because of such emissions. In addition it mentioned that besides GHG issues, oil palm plantations on peat also result in significant on- and off-site hydrological impacts such as soil subsidence and reduced water retention capacity. The resolution therefore called for the RSPO General Assembly to agree to establish a Committee, later known as the Peatland Working Group (PLWG) to explore and develop business models for optimising sustainability of existing oil palm plantations on peat, including options for restoration and after-use of peat, development of alternative economic uses, and application of water management regimes that lead to reduce emissions. The resolution was adopted by an overwhelming majority of RSPO members.

This report was commissioned by the RSPO PLWG and provides a review of available scientific information on the impacts of the use of tropical peat soils for oil palm cultivation in Southeast Asia. It assesses sources of uncertainty and gaps in knowledge, and structures the findings of available publications related to the cultivation of oil palm on tropical peat. In summary, the objectives of the review are:

- Examine the effects of establishing oil palm plantations on tropical peatlands on fluxes of CO_2 and other GHGs, and on other ecological, social, economic and livelihood issues.
- Define the spatial boundaries of the system and the major categories of GHG sources and sinks.
- Highlight uncertainties and gaps in knowledge.
- Provide recommendations for reducing GHG emissions and other adverse impacts.

Tropical Peatlands

The United States Department of Agriculture (USDA) defines peat as soils as histosols where more than half of the upper 100 cm consisting of organic matter. Peat is often also defined as a soil that contains at least 65% organic material, is at least 50 cm in depth, covers an area of at least 1 ha and is acidic in nature (Driessen, 1978; Wösten & Ritzema, 2001). The formation of peat depends on plant cover and hydrological conditions. Peat lands have their greatest extent in the boreal and temperate zones. Tropical peats are located in Southeast Asia, Africa, the Caribbean, and Central and South America and are also important components of the global terrestrial carbon (C) store in terms of both their above ground biomass (AGB) and their large underlying peat mass (Rieley *et al.*, 1996; Page *et al.,* 1999, 2004, 2011). Differences exist between peats in different climatic zones (Box 2). The most extensive tropical peat lands occur in Southeast Asia, representing 77% of the global tropical peat carbon store (Page *et al*., 2011b), most of which are located in Indonesia with 22.5 Mha (65% of global total of tropical peat) and Malaysia with about 2.4 Mha (10%) (Hooijer *et al.,* 2010). Awareness of the significant role that tropical peats and their forests play in the global carbon cycle has improved , and. while the full magnitude of this role is still uncertain (Malhi, 2010), recent studies have greatly increased our understanding of carbon emissions arising from peat land disturbance, especially for peat in Southeast Asia.

Box 2

Tropical lowland peat versus temperate and sub-arctic peat

Tropical lowland peat differs from temperate and sub-arctic peats. The latter are mainly derived from the remains of herbaceous plants (mainly species of *Sphagnum, Gramineae and Cyperaceae*) while tropical lowland peats are formed from the remains of woody forest species and, consequently, tend to have large amounts of undecomposed and partially decomposed trunks branches and woody roots that cause tropical peats to be formed at a much faster rates when compared to temperate peat bogs. Peats in cold and temperate regions are composed of humus-like compounds derived from decomposed cellulose, but peats in lowland swamp formations in tropical countries are composed largely of lignin, the compound that distinguishes wood from straw. Tropical peat soils decompose rapidly when exposed to aerobic conditions and drained peats usually consists of three horizons differentiated by their level of humification. The top or sapric horizon is most humified, followed by the hemic horizon (partially humified), while the bottom fibric horizon consists essentially of un-decomposed woody material.

Tropical peats in Southeast Asia occupy mostly in low altitude coastal and sub-coastal environments and extend inland for distances of hundreds of kilometres along river valleys and across watersheds. Most of these peatlands are located at elevations less than 50 m above mean sea level. Southeast Asian peats are largely ombrotrophic (receiving water by precipitation only), while a few basin peats are minerotrophic (receiving ground water and/or run off water) (Page *et al*., 2010). Peats occur along the coasts of East Sumatra, Kalimantan (Central, East, South and West), West Papua, Papua New Guinea, Brunei, Peninsular Malaysia, Sabah, Sarawak, Southeast Thailand and the Philippines, and can be subdivided into three main categories: 1) coastal, 2) sub-coastal or valley, and 3) high, interior or watershed (Rieley *et al*., 1996; Page *et al*., 1999, 2006). A combination of low topographic relief, waterlogged conditions, high effective rainfall and impermeable substrates provided conditions suitable for the accumulation of thick deposits of peat in these areas (Page *et al.,* 2010).

Information on peat structure, age, development and rates of peat accumulation is scarce. However, the study by Page *et al*. (2010) shows peat depth and carbon accumulation rates for four sites (in Peninsular Malaysia, in Kalimantan and in two areas in Sumatra), with depths ranging from 5.5 – 13.5 meters and accumulation rates ranging from 0 – 40 mm yr^{-1}. Peat accumulation occurs when the average rate of carbon sequestration exceeds the losses due to decomposition or runoff (Page *et al*., 2011b). Carbon content of tropical peat usually ranges between 40% and 60% depending on the nature, mineral content and location of the peat.

A study by Dommain *et al.* (2011) reported a mean Holocene carbon sequestration rate of 31.3 g C m^{-2} yr^{-1} for Central Kalimantan and 77.0 g C m^{-2} yr^{-1} for coastal sites in Indonesia, with the C content of the peat being 50-60% of its dry weight; a C content in line with results of studies by Neuzil (1997) and Page *et al.* (2004) in Central Kalimantan. The basic principle for the quantification of total organic carbon relies on the destruction of organic matter present in the soil. This can be performed chemically (the method often used in the past) or by using heat (the current method). In the studies where chemical methods were used, carbon contents were underestimated, giving values of 20-30% in tropical peat. Currently, the method using elevated temperatures is recommended.

The Peat Ecosystem

The carbon balance of tropical peat ecosystems is a result of CO_2 uptake by photosynthesis and release by respiration. The respiration component consists of heterotrophic respiration (decomposition of the peat by microbes) and autotrophic respiration (respiration from plant roots) (Page *et al*., 2011a). Besides their function as carbon sinks, tropical peat lands are unique ecosystems with a high biodiversity . Species diversity is regarded as one of the fundamental prerequisites of ecosystem stability. Until a few decades ago, tropical peat forests remained relatively undisturbed and acted as sinks for carbon. However, as a result of economic exploitation during the past two decades, peat swamp forests have been subject to intensive logging, drainage and conversion to plantations (Rieley & Page, 2002), and have thus been transformed into C sources.

Posa *et al.* (2011) state that the current extent and condition of tropical peatlands in Southeast Asia is still unclear, as accurate delineation of peat soil is difficult and many areas have already been lost or degraded. Using published estimates from various sources, they calculated the maximum remaining area of historical peat swamp forest to be 36.8% (*Table 1*).

The distribution of peat in Malaysia, Indonesia and Brunei in 2000 was determined by Wetlands International Malaysia (2010) using literature and satellite data (*Table 2*). In Malaysia, 7.5% of the total land area encompasses peat soils, of which Sarawak supports the largest area (69.1% of the total peat area in Malaysia), followed by Peninsular Malaysia (26.1%) and Sabah (4.8%) (Wetlands International, 2010). Wahyunto *et al.* (2005) reported that 10.8% of Indonesia's land area is comprised of peat lands, with Sumatra having 7.2 Mha, Kalimantan 5.8 Mha, Papua 7.9 Mha and other regions around 0.5 Mha. Page *et al.* (2010) have also published their best estimates of peat area, thickness and volume in Southeast Asia as shown in *Table 3*.

Table 1. Estimates of major peat swamp forest area (in ha) in SE Asia (Posa *et al.*, 2011).

Region	Initial Area (ha)	Remaining (ha)	% remaining	Protected (ha)	% Protected
Indonesia					
Sumatra	8,252,500	2,562,200	31.1	721,200	8.7
Kalimantan	6,787,600	3,160,600	46.6	763,200	11.2
Sulawesi	311,500	1,800	0.6	30,000	9.6
Malaysia					
Peninsular	984,500	249,200	25.3	44,400	4.5
Sabah and Sarawak	1,746,000	632,800	36.2	98,400	5.6
Brunei	104,000	87,300	83.9	21,800	21.0
Thailand	68,000	30,400	44.7	20,600	30.3
SE Asia Total*	**18,254,100**	**6,724,300**	**36.8**	**1,699,500**	**9.3**

*excluding Papua New Guinea

Table 2. The lowland peat extent in Southeast Asia and the estimated peat carbon stock, forest cover in 2000 and total area of degraded peatland using satellite data (Wetlands International Malaysia, 2010).

Country	Peat area (ha)	Peat carbon stock (Mton C)	Forested peatland in 2000 (ha)	Total degraded peatland area (ha)
Indonesia	26,550,000	54,016	14,000,000*	12,500,000
Brunei	99,100	98	85,000	14,000
Malaysia	2,668,500	5,431	140,000	1,200,000

**Bappenas estimated 14.000.000* ha *peat for Indonesia in 2009.*

Table 3. Best estimates of peat area, mean thickness and volume of peat in tropical Southeast Asia (Page *et al.*, 2010).

Country	Peat area (ha)	Average peat thickness (m)	Volume (m^3*10^6)
Indonesia	20,695,000	5.5	1,138,225
Brunei	90,900	7	6,363
Malaysia	2,588,900	7	181,223
Myanmar (Burma)	122,800	1.5	1,842
Papua New Guinea	1,098,600	2.5	27,465
Philippines	64,500	5.3	3,418.5
Thailand	63,800	1	638
Vietnam	53,300	0.5	266.5

Land Use Change and Deforestation

In Indonesia, peat development is most extensive in Sumatra, followed by Kalimantan; most of the peat formations in Papua remain undeveloped. In Malaysia, deforestation rates in the past 6 years were highest in Sarawak with a yearly deforestation rate of around 8% on average for peat land (SarVision, 2011; *Table 4a*), and an overall deforestation rate of around 2% in the last 5 years for all soil types (SarVision, 2011; *Table 4b*).

Table 4a. Yearly deforestation of peatland in Sarawak,Malaysia in the period 2005-2010 (SarVision, 2011)

Year	Forest area (ha)	Forest area change (ha)	% change
2005	1,055,896.7	No data	No data
2006	990,437.6	-65,459.1	-6.20
2007	924,978.5	-65,459.1	-6.61
2008	847,256.4	-77,722.1	-8.40
2009	769,534.3	-77,722.1	-9.17
2010	702,966.7	-66,567.5	-8.65

Table 4b. Yearly total deforestation in Sarawak, Malaysia in the period 2005-2010 (SarVision, 2011).

Year	Forest area (ha)	Forest area change (ha)	% change
2005	8,984,450.7	No data	No data
2006	8.814,801.7	-169,648.9	-1.89
2007	8,645,152.8	-169,648.0	-1.92
2008	8,470,649.8	-174,503.0	-2.02
2009	8,296,146.8	-174,503.0	-2.06
2010	8,118,614.4	-177,532.4	-2.14

Table 5 lists studies on peat swamp forest loss for different areas in Southeast Asia. Overall, deforestation rates in Sarawak, Malaysia are the highest and SarVision (2011) reported that 41% of the peat soil in Sarawak was covered by oil palm plantations by 2010. In a study by Miettinen *et al.* (2011), deforestation rates in insular Southeast Asia were determined by comparing satellite imagery between 2000 and 2010 using a spatial resolution of 250 m to produce land cover maps using regional classification schemes (*Table 6*). The results revealed an overall 1.0% yearly decline in forest cover when considering Brunei, Indonesia, Malaysia, Singapore and Timor Leste, of which 68%-80% of the total study area was turned into plantations or underwent regrowth (shrub land to young secondary forest). In the past years, deforestation rates for peat swamp forest were higher than deforestation rates for forests on mineral soils.

By excluding Papua and the Moluccas from the analysis, the yearly rate of forest loss for Indonesia rises to 1.5% (3.3% for peat swamp forest). The highest deforestation rates were found for the eastern lowlands of Sumatra (mainly Riau and Jambi provinces) and for the peat lands of Sarawak. In both of these areas deforestation was concentrated in peat lands. Riau and Jambi provinces together had lost 40% by the peat swamp forest cover by 2010, while in Sarawak the extent of peat swamp forests decreased by 55% (Miettinen *et al.*, 2011). Earlier studies of these areas reported average yearly deforestation rates of 1.7% between 1990-2000 (FAO, 2006), 2.0% between 1997-2002 for Borneo (Fuller *et al.*, 2004) and 1.5% between 1990-2000 for Indonesia (Hansen *et al.*, 2009).

Miettinen *et al.* (2012) did an extensive study using high-resolution satellite imagery to analyse sequences and interrelations in the progression of peat degradation and conversion processes in Sumatra, Indonesia (*Table 7*). Changes were monitored in three study areas of 2,500–3,500 km^2 since the 1970's and examined in conjunction with satellite-based active fire data sets. They concluded that forests disturbed by intensive logging and/or drainage are merely intermediate stages leading to further change, such as plantation establishment.

Table 5. Peat swamp forest loss (%) for different areas in Southeast Asia, for different periods in time.

Area	Period	Reference	Peat swamp forest converted to other LU % of peat forest (average)
Insular SE Asia	2000-2005	Wetlands International Malaysia 2010	1.47
Sarawak	2005-2007	SarVision 2011	7.1
Sarawak	2009-2010	SarVision 2011	8.9
Malaysia and Indonesia	2000-2010	Miettinen *et al* 2011	2.2
Borneo	1997-2002	Fuller *et al* 2004	2
Indonesia	1990-2000	Hansen *et al* 2009	1.5

Table 6. Forest cover change from 2000-2010. Peat swamp forest numbers are given in Italics (Miettinen *et al.*, 2011)

Area	2000		2010	
	x 1000 ha	%	x 1000 ha	%
Peninsular Malaysia	5,388	41.1	4,947	37.7
	287	***2.2***	***235***	***1.8***
Sumatra	14,555	33.5	11,104	25.5
	3,131	***7.2***	***1,839***	***4.2***
Borneo	41,688	56.6	36,688	49.8
	4,182	***5.7***	***3,144***	***4.3***
Java	866	6.8	902	7.1
	0.0	***0.0***	***0.0***	***0.0***
Sulawesi	8,959	53.0	7,993	47.1
	0.0	***0.0***	***0.0***	***0.0***
New Guinea	31,625	84.4	30,859	82.7
	6,336	***17.0***	***5,970***	***16.0***
Indonesia	94,867	51.3	86,039	46.5
	12,740	***6.9***	***10,541***	***5.7***
Malaysia	17,242	52.4	14,962	45.4
	1,230	***3.7***	***673***	***2.0***
Total study area	112,536	51.2	101,434	46.1
	13,970	***6.4***	***11,214***	***5.1***

Table 7. Land cover changes in the study areas (1970's – 2009/2010) in Sumatra (Miettinen *et al.*, 2012).

Land Cover	North Sumatra		Riau		Jambi			
					Outside Berbak nat. park		Inside Berbak nat. park	
	1977	2009	1979	2010	1970's	2009	1970's	2009
Nearly pristine forest	190.8	0	202.4	5.56	183	53.1	120.1	92.2
Moderately Degraded forest	14.6	2.9	0.6	2.23	8.2	14.9	5.3	5.5
Heavily Degraded forest	0.6	11.1	0	7.5	2.4	29.3	0	1.8
Secondary forest	4.6	5.1	0	1.8	0.1	18.0	0	5.2
Clearance/burnt	0	10.8	0	12.6	0	4.3	0	1.1
Smallholder mosaic	10.7	69.1	7	11.9	7.3	17.6	0.1	0.7
Industrial plantation	1.9	87.9	0	6.07	0	27.9	0	0

Areas are given in ha x 10^3

Plantation Development

Oil palm (*Elaeis guineensis*) has become one of the most rapidly expanding food and biofuel crops in the world. The two main palm oil producing countries are Malaysia and Indonesia, with Malaysia currently responsible for up to 38% and Indonesia for up to 49 %, of the world's palm oil production (Figure 1).

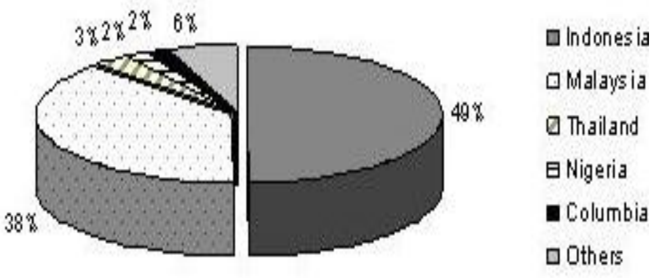

Figure 1. World palm oil production in 2010. (see www.indexmundi.com/agriculture).

A large part of the area needed for the expansion of the palm oil industry has involved the conversion of forest. A study by Wicke *et al.* (2008) shows that in Indonesia the largest land use change was from forest to oil palm and other agricultural crops, while in Malaysia oil palm development has been mainly at the expense of other permanent crops, rather than directly from deforestation. The causes of forest cover loss in Malaysia vary with region. In Sabah and Sarawak, the most important causes have been timber extraction and shifting cultivation, while in Peninsular Malaysia, and in recent years increasingly in Sabah, forest cover has been affected most by direct conversion to agriculture and more specifically to oil palm plantations (Wicke *et al.*, 2010). The largest change in Indonesia has occurred in forested land, which decreased from 130 Mha in 1975 to 91 Mha in 2005, while agricultural land increased from 38 Mha in 1975 to 48 Mha in 2005. Approximately half of this agricultural expansion was due to an expansion in palm oil production (Wicke *et al.*, 2010).

A recent study documented oil palm land use in Malaysia (Peninsular Malaysia, Sabah and Sarawak) using 2008-2009 satellite images (Omar *et al.*, 2010). The total area of oil palm detected was 5.01 Mha, of which 0.67 Mha was on peat (*Table 8*). According to this study, the largest proportion (>37%) of oil palm plantations on peat in Malaysia, some 0.44 Mha, occurred in Sarawak. In Indonesia, oil palm plantations on peat are currently estimated to cover 1.3 Mha, with around 1.0 Mha in Sumatra and 0.3 Mha in Kalimantan (Page *et al.*, 2011a,b). *Table 9* shows the area of oil palm concessions on peat (which represent future development) to increase to a total of 2.5 Mha in Sumatra and Kalimantan by 2020 (Hooijer *et al.*, 2006; Page *et al.,* 2011a,b).

Table 8. Oil palm on peat in 2009 Malaysia (Omar *et al.*, 2010).

Region	Oil Palm (ha)	Oil Palm on peat	
		(ha)	(%)
Peninsula Malaysia	2.503.682	207.458	8.29
Sabah	1.340.317	21.406	1.60
Sarawak	1.167.173	437.174	37.45
Total	**5.011.172**	**666.038**	**13.29**

Table 9. Oil palm concessions (projections 2020) on peat in 2006 in Indonesia (Peat-CO_2 report Wetlands International, 2006, by SarVision).

Region	Peat Area (ha)	Oil Palm plantation concessions on peat (ha)	Percentage of peat with oil palm plantation concessions (%)
Sumatra	6.931.700	1.249.400	18
Kalimantan	5.837.900	1.472.500	25
Papua	7.554.300	79.000	1
Total	**20.323.900**	**2.800.900**	**14**

Several studies have been performed based on past trends, land availability and projected demand for palm oil. These calculated the possible expansion of oil palm, 1) according to past land use change trends (*business as usual*), 2) using all available land to grow oil palm (a *maximum* production scenario), and 3) a scenario emphasising sustainability criteria (*sustainable* case). The most sustainable scenario avoids the use of forest land, steep terrain, and vulnerable peat soils for oil palm plantation establishment (Kaper *et al.*, 2008). Wicke *et al.* (2008) and Germer & Sauerborn (2006) concluded in their studies that in order for oil palm products to be sustainably produced, only non-peat, low-carbon, degraded land should be used for palm oil production and plantation management should be improved. With growing demand for both food and fuel for export, as well as for domestic biodiesel production, it is likely that significant further land use conversions to oil palm will occur (Koh & Wilcove 2007) and this will put further pressure on peat swamp forest ecosystems (Rijenders & Huijbregts, 2008; Fargione *et al.*, 2008). While biofuels such as palm oil were identified initially as potential low-carbon energy sources, further research has shown that oil palms grown on peat create a 'carbon debt' and so increase overall global carbon emissions (Fargione *et al.*, 2008; Gibbs *et al.*, 2008).

Implications of Land Use Change

Carbon and greenhouse gas implications

Tropical peat swamp forest ecosystems are one of the most important terrestrial carbon stores on earth. Indonesian peat lands store at least 55 ± 10 Pg (gigaton) of carbon, equal to 10-30% of the global peat carbon stock (Jaenicke *et al.*, 2008; Page *et al.*, 2002) and Malaysian peats store around 9 Pg of carbon (Page *et al.*, 2011b). The most important factor that controls the peatland C-balance is hydrology (Jauhiainen *et al.*, 2005; Couwenberg *et al.*, 2010). Drainage of peat leads to peat oxidation and a higher frequency of fires, resulting in an increase in GHG emissions and carbon loss (Gomeiro *et al.*, 2010). Conversion of forest for agricultural development is a one-point emission in time, while emissions resulting from peat drainage are continuous processes. Emissions due to peat drainage are not caused just by *land use change*, which generally involves a loss of biomass, but rather to its long-term effects on the carbon store in the soil. This is different in the case of deforestation on mineral soils, where the largest proportion of emissions results from the loss of biomass at the time of land use change.

Other ecological implications

The rapid and massive expansion of oil palm has also led to concerns about its impact on natural habitats and biodiversity (Fargione *et al.*, 2008; James, 2008; Koh & Ghazoul, 2008). Locally, the development of oil palm plantations in forested areas will have several consequences, such as increased erosion, loss of biodiversity, pollution by chemical runoff, and increased fire risk (Naidoo *et al.*, 2009). Other impacts include soil subsidence due to drainage and fires, which can lead to an increased risk of flooding, salt water intrusion, and, in some cases, eventual loss of the entire peat formation. Oil palm monocultures require use of insecticides, herbicides and fertilizers, which may enter water bodies as runoff or groundwater seepage and can seriously impact aquatic biodiversity (Koh & Wilcove, 2008). Another problem is haze following peat and forest fires. Exposure to high levels of air pollution increases risk of asthma, bronchitis and other respiratory illnesses (e.g. Brown 1998; Sastry 2000).

Haze can also result in the reduction, by as much as 92%, in photosynthetically active radiation (PAR) which can affect rates of carbon fixation (Yule, 2010).

Social, economic and livelihood implications

The broader economic, social and livelihood implications of oil palm cultivation on peat remain poorly understood (Rist *et al*., 2009; Rist *et al*., 2010). Although many households profit from the palm oil business, the expansion of large-scale oil palm plantations will lead to loss of ecosystem services. Some studies warn of instability in food prices because smallholders may become over dependent on the price of palm oil. In Indonesia, one point of concern is from transnational corporations and other large landowners who establish extensive landholdings at the expense of small farmers (Rist *et al*., 2010). However, many findings are contradictory and differ among regions and may be affected by the time frame of the studies, while short term economic consequences are often positive, the longer term implications can be the reverse. Figure 2 shows the linkages that exist between the loss of peat swamp forests and global market forces, as mediated by national export policies and international investments. The increasing demand for a product in one part of the world may negatively impact wetland ecosystems elsewhere. In the process, the conservation and sustainable management of tropical peats in Southeast Asia is threatened. Nonetheless, oil palm appears to be an attractive new income opportunity for Indonesian farmers, as attested by its widespread uptake by many smallholder communities (Rist *et al*., 2010). Oil palm is widely considered by these communities as the best option for reducing rural poverty.

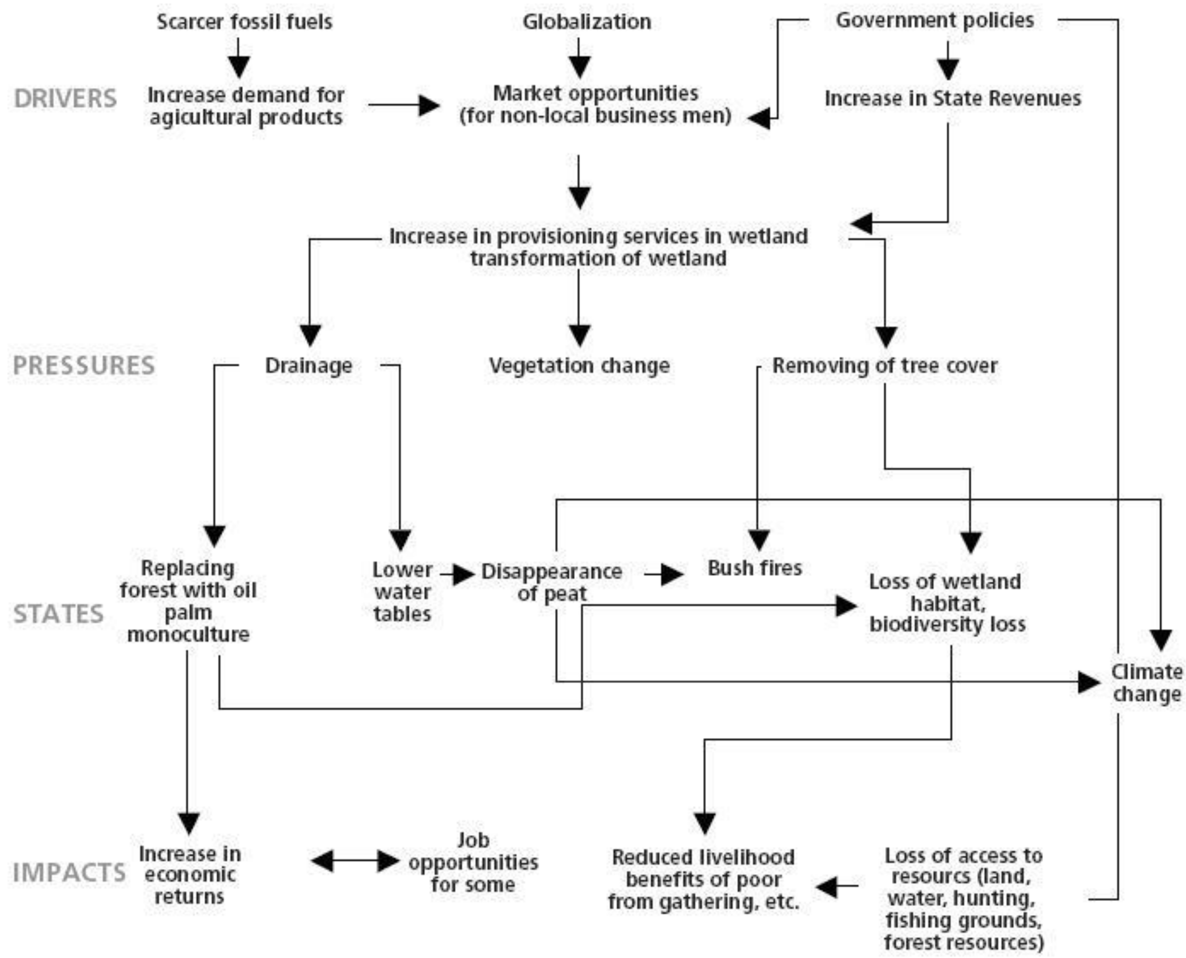

Figure 2. Transformation of wetlands in perspective: schematic overview of drivers, pressures, states and impacts (FAO, 2008). Note that the increased demand for palm oil as food is not included in this scheme.

CARBON BALANCES AND GREENHOUSE GAS EMISSIONS IN TROPICAL PEATLANDS

Introduction

Intact peat swamp forests store large amounts of carbon in the peat and in the vegetation. Since the 1980s large areas of tropical peat swamp forest in Southeast Asia have been converted for urban development, forestry and agriculture, including for palm oil production. Conversion of tropical peat forest areas into agricultural land has various consequences for the carbon and GHG balance in the years following disturbance. These consequences are mainly dependent on the extent of deforestation, drainage depth and water management.

Data Availability and Restrictions

Although a lot of research has been performed in the past, using different approaches (Box 3), some of the earlier studies on GHG fluxes suffered from several methodological limitations. General pitfalls were:

- The short-term nature of the studies (usually < 1 y), with a limited number of point measurements over time. In the tropics, large differences in annual balances can be expected between dry and wet years.
- Failure to address temporal and spatial variability in a systematic way.
- Use of linear interpolation to perform temporal upscaling of fluxes instead of using a regression based approach.
- The focus of most studies on CO_2 with relatively few studies on other major GHGs such as N_2O (which arises from fertilizer applications) and CH_4 (a potential emission source fom drainage ditches).

Comparison studies of CO_2 emissions have largely been based on chamber measurements of total soil respiration and have failed to distinguish between autotrophic and heterotrophic respiration (Melling *et al*., 2005b; Melling *et al*., 2007; Furukawa *et al*., 2005; Reijnders & Huijbregts, 2008; Hadi *et al*., 2005).

Flux estimates can also be seriously biased by the failure to detect and allow for 'event' emissions such as those due to sudden climatic changes or discontinuous management activities, such as changes in temperature or rainfall, fertilizer application, and dredging (Kroon *et al*., 2010; Veenendaal *et al*., 2007; Hirano *et al*., 2007). Atypical results, or outliers, may be caused by pressure changes during chamber installation, which results in very high fluxes that can dominate the overall balance estimate. A complex micro-topography may be present consisting of hummocks and hollows than can cause a spatial bias, which may not be representative of the total area.

Studies have been undertaken in the last few years that avoid or minimise these potential problems. One approach is to collect data from several studies and attempt to infer emissions based on drainage depth (Couwenberg *et al*., 2010). Others have tried to avoid all major deficiencies related to chamber measurements (Jauhiainen *et al.*, 2012). Some studies base their carbon and CO_2 emission estimates on soil subsidence rates (Dradjad *et al*., 2003; Couwenberg *et al*., 2010; Hooijer *et al*., 2012), using an assumed bulk density and allocating a percentage of subsidence to peat oxidation. The latest methods for calculating CO_2 emissions from soil subsidence avoid the use of an assumed oxidative component by using the bulk density of peat below the water table as a proxy for the original bulk density of the peat above the water table, which integrates the impact of initial consolidation. Compaction continues to work on consolidated peat, however, once it reaches the aerated zone above the water table (Couwenberg & Hooijer, 2013).

Ecosystem flux values differ depending on the system boundaries. Some studies address the entire oil palm biofuel production chain; others include management-related fluxes or only soil respiration within a single plantation (Figure 3). The amount of release or uptake of GHGs in an ecosystem is dependent on a variety of interrelated processes, including climate and variables such as temperature, moisture, water table depth, microbial activity, drainage, logging, compaction, peat type, and vegetation type. To completely understand the temporal and spatial variation of fluxes from a peat ecosystem and to upscale fluxes from a small (m^2) to the landscape scale, these processes and variables and their inter-relationships need to be documented.

Box 3

Greenhouse gas and carbon measurement techniques

Chamber based methods: Sample areas are usually smaller than one square meter (1 m^2) and are discontinuous in both space and time. They are best suited for capturing spatial variability and can be used to measure fluxes of the three major GHG: CO_2, CH_4 and N_2O. If appropriate, spatial and temporal upscaling methods can be used to determine average GHG fluxes at the landscape scale (note: correct spatial stratification requires regression analyses rather than a simple linear interpolation).

Eddy covariance (EC) based methods: These cover areas of 100 – 1000 m^2 depending on the height of the measurement instruments, which are located mainly on towers that extend above the vegetation canopy. An array of instruments on these towers continuously measure both incoming and outgoing radiation, GHG fluxes, and energy exchanges. The EC technique is best suited for determining average GHG fluxes at the landscape scale and for capturing temporal variability at multiple temporal scales ranging from a single day to several months or even years. EC techniques for CO_2 have been used for more than a deade, while EC techniques for CH_4 and N_2O are still under development. The EC technique integrates emissions over large areas, and footprint analysis (models used to estimate where the fluxes originate) is currently insufficient to capture small scale variability.

Soil subsidence based methods: In principle, land subsidence can be determined using several straight forward measurement techniques, such as leveling surveys, subsidence poles and Global Positioning System (GPS) systems. A field study in Johor, Malaysia determined the oxidation component of subsidence to be about 60% (Wösten *et al.*, 1997), but other studies based on several large-scale studies in subtropical and tropical regions have estimated the oxidative component of subsidence to be around 90% (Stephens *et al*., 1984; Hooijer *et al*., 2012). Recently, soil subsidence methods avoid the errors in estimating the oxidative component by using the bulk density of the peat below the water table as a proxy for the original bulk density of the peat above the water table (Couwenberg & Hooijer, *2013*).

Satellite based approaches: These usually focus on loss of carbon by documenting land use change and deforestation at relatively large scales. Changes in soil carbon stocks, in both mineral and peat soils, are usually not included in these studies, except by the use of models based on assumptions derived from ground-based studies. Satellites are extremely useful, however, for monitoring the distribution and frequency of fires, which can be used for estimating carbon loss from peat fires.

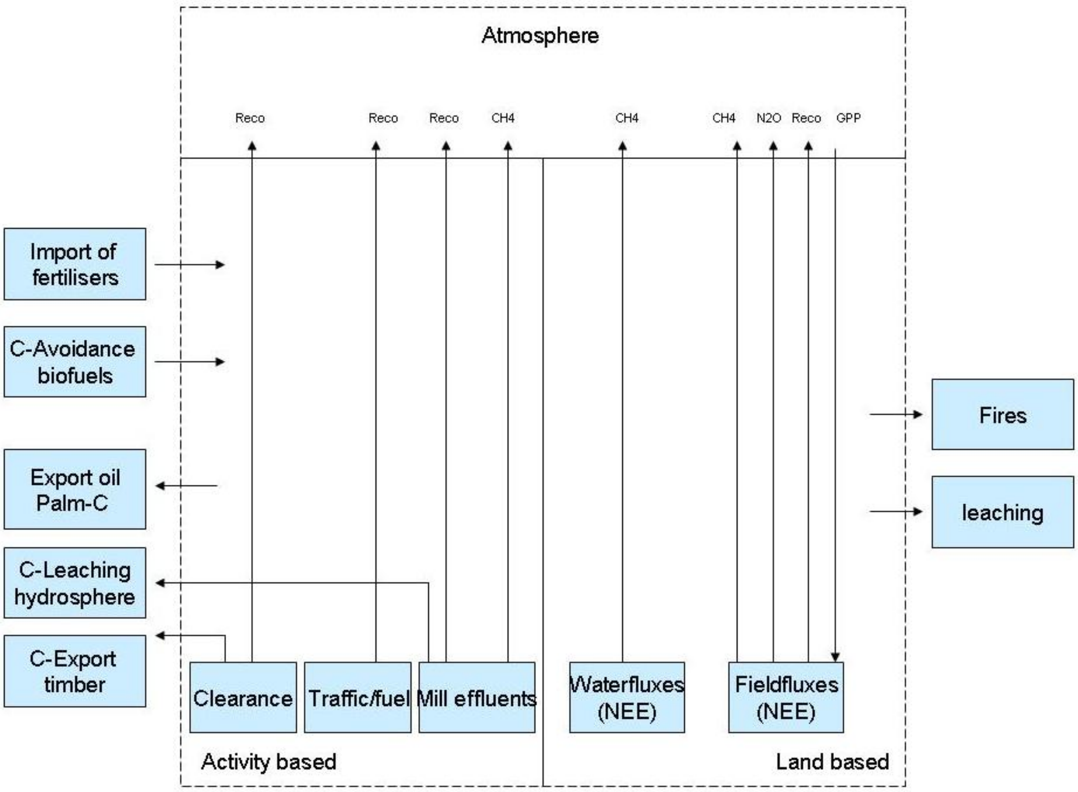

Figure3. System boundaries of an oil palm plantation (dotted line) with the carbon (C) and GHG sources and sinks: NEE = Net Ecosystem Exchange, GPP = gross primary production or photosynthesis, Reco = ecosystem respiration, CH_4 = methane, N_2O = nitrous oxide, CO_2 = carbon dioxide.

Carbon Dioxide and Carbon

Direct loss of carbon

Agricultural development of tropical peat involves a change in vegetation cover and, in almost all cases, permanent drainage. The land use change from forest to oil palm plantation (clearing and/or burning of AGB), causes a direct loss of carbon (Danielsen *et al*., 2009) ranging from 111-432 Mg C ha^{-1} in natural or primary peat swamp forest to 73-245 Mg C ha^{-1} in logged forest, while the carbon stock in oil palms ranges from only 25-84.6 Mg C ha^{-1} (Agus *et al*., 2009; Lasco, 2002; Gibbs *et al.,* 2008; Verwer & van der Meer, 2010; Murdiyarso *et al.*, 2010). Loss of forest cover in Southeast Asia can be grouped into three main categories: 1) forest degradation caused by logging, 2) conversion of forest areas into large scale plantations by clear felling, and 3) expansion of small-holder dominated farming areas (Miettinen *et al*., 2011). The effects of logging may be highly variable depending on logging intensities, rotation cycles and damage to the residual stand. Root biomass in relatively undisturbed peat swamp forests is estimated at 29-45 Mg C ha^{-1} (Verwer & van der Meer, 2010) and can be a further source of carbon loss following conversion.

CO_2 emissions from land use change

Deforestation

Forests absorb CO_2 by photosynthesis and release it by respiration; autotrophic respiration refers to the respiration from roots and above ground plant organs. Soil respiration is the CO_2 release at the soil surface due to microbial activity, referred to as heterotrophic respiration, and the autotrophic respiration of plant roots. Suzuki *et al.* (1999) demonstrated in their micrometeorological studies in tropical peat forest in Thailand that 5.32 Mg C ha^{-1} yr^{-1} was absorbed by the primary peat swamp forest canopy in photosynthesis while secondary forest absorbed 5.22 Mg C ha^{-1} yr^{-1} because of greater plant growth compared to primary forest. During deforestation for development of an oil palm plantation, living biomass is harvested; at the same time, gross primary production (GPP) decreases and the net ecosystem exchange (NEE) increases (Hirano *et al*., 2007). The carbon loss from forest conversion exceeds the potential carbon fixation of oil palm plantings and, in addition, artificial drainage needed for cultivation of oil palm on peat will increase microbial respiration compared to the situation without drainage (e.g. Jauhiainen *et al.,* 2005; de Vries *et al.,* 2010; Henson, 2009; Jeanicke *et al.,* 2008; Danielsen *et al.,* 2009; Fargione *et al.,* 2008; Rieley *et al*., 2008; Gibbs *et al*., 2008; Wösten & Ritzema, 2001; Hooijer *et al.,* 2006).

Drainage

Drainage causes peat carbon to be oxidised and released as CO_2. It also increases the risk of peat fire (Furukawa *et al.,* 2005; Wösten *et al*., 1997; Inubushi *et al*., 2003; Hooijer *et al*., 2006; Veenendaal *et al*., 2007). Page *et al.* (2011a) concluded that a value of 86 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} represents the most robust, empirical estimate of peat CO_2 emissions currently available for oil palm plantations on deep, fibric peat with uncertainties ranging from 54 to 115 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for typical drainage depths of 60 – 85 cm, when annualized over 50 years and including the initial emission peak just after drainage. Couwenberg & Hooijer (2013) suggest a CO_2 emission value of 55-73 Mg $CO_{2\text{-eq}}$ ha^{-1} yr^{-1} for continuous peat emissions under best to common practice, management, excluding initial emissions just after drainage. Couwenberg *et al.* (2010) and Hooijer *et al.* (2010) calculated emissions of at least 9 Mg CO_2 ha^{-1} yr^{-1} and 9.1 Mg CO_2 ha^{-1} yr^{-1}, respectively, for each 10 cm of additional drainage depth. Transforming an undrained peat with the water table at the soil surface into a drained peat area with a drainage depth of 60-80 cm would thus increase the peat emissions by about 55-72 Mg CO_2 ha^{-1} yr^{-1} (Figure 4).

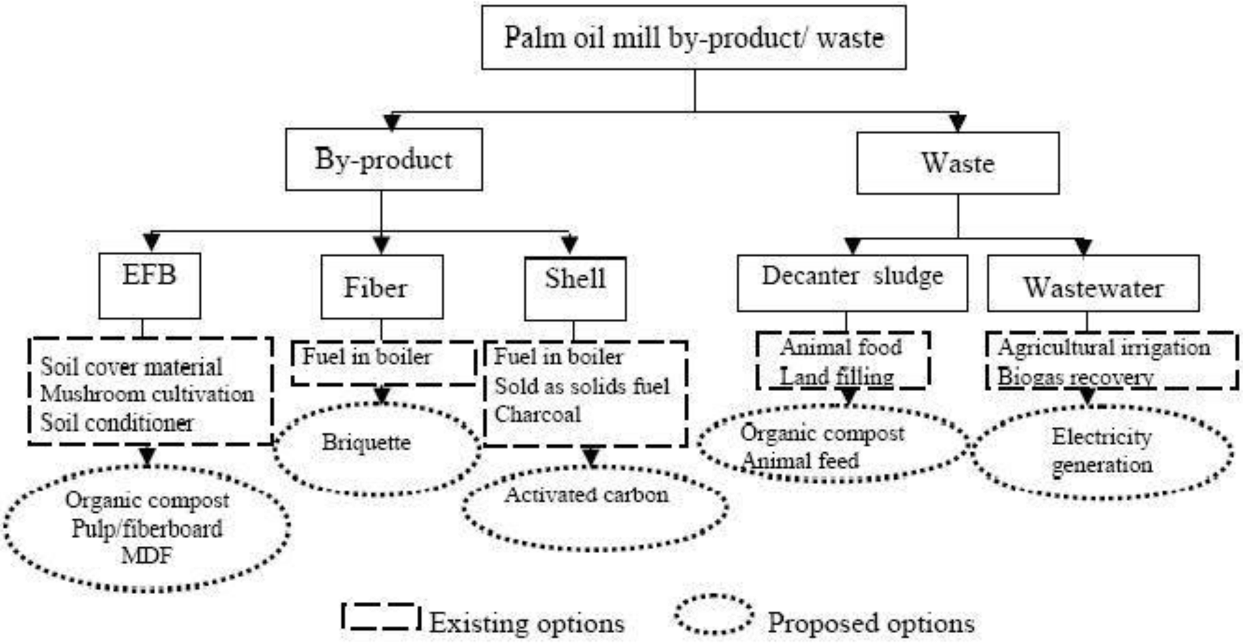

Figure 4. By-products and wastes from oil fresh fruit bunch (FFB) processing (Chavalparit, 2006).

These relations have been refined recently as more field data have become available (Hooijer *et al.*, 2012; Jauhiainen *et al.*, 2012) both from subsidence studies that account for changes in bulk density (thus correcting for compaction and consolidation), and from CO_2 gas flux measurements that exclude root respiration.

Recent studies showed that emissions in both *Acacia* and oil palm plantations after more than 5 years following initial drainage (i.e. excluding the initial peak) was consistently around 73 Mg CO_2 ha^{-1} yr^{-1} with a water table depth of 0.7 m. Note that the initial peak may be as high as 178 Mg CO_2-eq ha^{-1} yr^{-1} in the first 5 years after drainage (Hooijer *et al.*, 2012). Page *et al.* (2011a) after reviewing available literature concluded that around 73 Mg CO_2 ha^{-1} yr^{-1} is released from drained peat in oil palm plantations, but increases to 86 Mg CO_2 ha^{-1} yr^{-1} if the initial peak directly after drainage is taken into account. Lower estimates were found by Melling *et al.* (2005a) who reported a value of 55 Mg CO_2 ha^{-1} yr^{-1}. It should be noted that studies in Sarawak, such as those by Melling *et al.* (2005a), reflect a different rainfall regime than those in most of Indonesia, where dry season rainfall is lower, soil moisture deficits are common; consequently, the rate of peat oxidation and carbon loss are expected to be substantially higher. The most recent research proposes a mean CO_2 emission rate of 64 Mg CO_2 ha^{-1} yr^{-}1, with a range between 55-73 Mg CO_2 ha^{-1} yr^{-} for continuous peat emission, excluding the initial peak (Couwenberg & Hooijer, 2013). This is in line with the previous equations by the same authors of ~ 9 Mg CO_2-eq ha^{-1} yr^{-}1 per 10 cm of drainage depth.

One of the few studies in Indonesia and Malaysia that used the eddy covariance methodology to measure fluxes in a degraded and drained tropical peat swamp forest using the total CO_2 balance approach (Hirano *et al.* 2007) showed that the drained forest appeared to be a CO_2 source of 16 Mg CO_2 ha^{-1} yr^{-1}, which was the difference between the uptake by living biomass (GPP) of 126 Mg CO_2 ha^{-1} yr^{-1} and an ecosystem respiration (Reco) of 142 Mg CO_2 ha^{-1} yr^{-1}.

In tropical regions, peat oxidation is dependent on factors such as time of year (dry-wet season), quantity and quality of organic matter, and environmental factors such as soil temperature and moisture (e.g. Hirano *et al.*, 2007). Even in the small range of temperatures typical for tropical areas, particularly in the early stages of plantation establishment when the canopy is not closed, emissions are positively related to temperature (Hooijer *et al.*, 2012; Jauhiainen *et al.*, 2012; Murdiyarso *et al.*, 2010; Hirano *et al.*, 2007).

Fires

An indirect result of drainage and inappropriate management activities is an increase in fire frequency (Hope *et al*., 2005). Although land clearance by fire has been banned for several years, it is still a widespread practice, particularly by smallholders who lack access to heavy machinery (Page *et al*., 2011a). Couwenberg (2010) estimated a release of 260 Mg C ha^{-1} yr^{-1} during the 1997 peat fires in Southeast Asia, which corresponded well with the estimates of van der Werf *et al*. (2008) and Page *et al.* (2002). Limin *et al.* (2004) estimated a carbon emission of 186 and 475 Mg ha^{-1} respectively for the drought years 2002 and 1997. Based on available measurement data, the mean burn depth and rate of fire related peat loss amounted to 34 cm per fire event and 261 Mg C ha^{-1} yr^{-1} averaged for the years 1997, 2001 and 2002 in an abandoned, degraded peat area (Heil, 2007). Additionally, the ash produced during a fire enhances peat decomposition (Murayama & Bakar, 1996).

Other CO_2 emission sources

The focus of this chapter is on emissions from peat; however, to create a complete and clear picture of the system as shown in Figure 3, management related fluxes also have to be taken into account. Oil processing leads to losses of carbon and GHGs because mills produce large amounts of organic waste. These losses add to the emissions for oil palm plantations on peat soils, as well as those on mineral soils. Figure 4 shows the wastes from fresh fruit bunches (FFB) as studied by Chavalparitk (2006). Data from Thai production for 1993 suggests that on a weight basis such wastes amount to nearly 80% of the inputs (Prasertsan *et al*., 1996). Based on the OPCABSIM model of Henson (2009) (RSPO, 2009), C losses through fossil fuel use were estimated to be 0.39 Mg C-eq ha^{-1} yr^{-1} (1.43 Mg CO_2 ha^{-1} yr^{-1}), losses through initial biomass loss (e.g. FFB waste) were 3.47 Mg C-eq ha^{-1} yr^{-1} (12.7 Mg CO_2 ha^{-1} yr^{-1}) and carbon gains through fertilizer inputs were 1.5 – 2 Mg CO_2 ha^{-1} yr^{-1}.

The drainage needed for the cultivation of oil palm means that dissolved organic matter leached to drainage ditches and rivers will also be enhanced (Rixen *et al*., 2008; Miyamoto *et al*., 2009; Yule & Gomez, 2009), especially in the transitions from dry to wet periods. Increases of 15% in dissolved organic carbon have been recorded during this transition (Rixen *et al*., 2008). The carbon exported rapidly decomposes, causing high fluxes of CO_2 from water bodies (Couwenberg *et al*., 2010; Holden *et al*., 2004). A recent study concluded that the fluvial organic carbon flux from disturbed, drained peat swamp forest is about 50% larger than that of undisturbed peat swamp forest (Moore *et al*., 2013). These workers concluded that adding these fluvial carbon losses (estimated at 0.97 Mg C ha^{-1} yr^{-1}) to the total carbon budget of disturbed and drained peatlands increased the total ecosystem carbon loss by up to 22%. Jauhiainen & Silvennoinen (2012) used floating closed chambers to measure GHG fluxes from drainage ditches in tropical peatlands, including plantations, and found that total GHG fluxes from canals are generally higher than from the neighbouring fields. They found fluxes of 15.2 Mg CO_2-C ha^{-1} yr^{-1} from drainage ditches in disturbed peat areas (with a ditch area 2% of the total), which is in the same order as the fluxes found by Moore *et al*. (2013).

Methane

Methane is formed from organic or gaseous carbon compounds by methanogenic bacteria living in the anaerobic, water saturated peat layers. In the upper, more oxic peat layers methanotrophic bacteria oxidize part of the CH_4, diffusing it upwards as CO_2. Currently it is believed that the emissions of CH_4 from tropical peat areas only make a minor contribution to the GHG flux compared to the emissions of CO_2, and thus play only a minor role in the carbon balance. However, the extent of emissions from open water and those promoted by management practices and fires, are likely to contribute considerably, particularly because the warming potential of CH_4 is 25 times that of CO_2. However, net CH_4 fluxes from tropical peats are low compared to fluxes from temperate peat soils and they usually show a clear positive relationship to water level for water levels above 20 cm, as is also the case for temperate wetlands (Watanabe *et al*., 2009). An overview of the available scientific literature on methane emissions in tropical peat is given in *Table 10* and Appendix A.

Table 10. Annual terrestrial (land based) methane emissions from peat in tropical Southeast Asia from available scientific literature calculated in different ways. Fluxes related to open water and to management activities are excluded.

Reference	Land use	Chamber measurements frequency	Mean CH_4 emissions (g CH_4 m^{-2} yr^{-1})	Min CH_4 emissions (g CH_4 m^{-2} yr^{-1})	Max CH_4 emissions (g CH_4 m^{-2} yr^{-1})	Mean CO_2-eq (t CO_2 ha^{-1} yr^{-1})	Min CO_2-eq (t CO_2 ha^{-1} yr^{-1})	Max CO_2-eq (t CO_2 ha^{-1} yr^{-1})
Ueda *et al*, 2000	Fresh water swamp			4.38	109.5		1.05	26.28
Hadi *et al*, 2005	Rice	1 year, monthly		3.5	14.0		0.3	1.22
	Sec. forest	1 year, monthly	5.87			1.41		
	Paddy field	1 year, monthly	26.13			6.28		
	Rice-soybean	1 year, monthly	3.47			0.83		
Couwenberg *et al*, 2010*	Swamp forest	1 year, monthly on average		-0.37	5.87		-0.9	1.41
	Agriculture	1 year, monthly on average		0.025	3.4		0.006	0.816
	Rice	1 year, monthly on average		3.26	49.5		0.87	11.88
Melling *et al*, 2005	Sec. forest	1 year, monthly	0.02			0.006		
	Sago	1 year, monthly	0.24			0.06		
	Oil palm	1 year, monthly	-0.02			-0.006		
Furukawa *et al*, 2005	Drained forest	1-2 years, monthly	1.17			0.28		
	Cassava	1-2 years, monthly	3.39			0.81		
	Paddy field upland	1-2 years, monthly	3.62			0.87		
	Paddy field lowland	1-2 years, monthly	49.52			11.89		
	3 Swamp forests	2 months	6.15			2.02		

* Combined research adapted from Couwenberg *et al.*, 2010: Inubushi *et al.*, 2003; Furukawa *et al.*, 2005; Hadi *et al.*, 2005; Jauhainen *et al.*, 2005; Melling *et al.*, 2005; Takakai *et al.*, 2005; Hirano *et al.*, 2009.

CH_4 emissions from land use change

Only a few studies have focused on CH_4 fluxes from tropical peat land. Couwenberg *et al.* (2010) concluded that CH_4 emissions in tropical peat are negligible at low water levels and amount to up to 3 Mg CH_4 m^{-2} hr^{-1} (6.3 kg CO_2-eq ha^{-1} yr^{-1}) at high water levels. Raised soil temperature following land use change may stimulate the process of methanogenesis, and the abundance of drainage canals, ponds or flooded areas may promote CH_4 emissions to non-negligible levels (Jauhiainen *et al.*, 2012). In some temperate regions, these emissions from water bodies may account for 60% of the total annual CH_4 flux of a drained peat ecosystem, depending on the amount of nutrients in the water and its depth (Schrier-Uijl *et al.*, 2011). Typical drainage parameters , such as the spacing and width of canals, in oil palm plantations in Indonesia (Table 10) show that water surface from drainage canals may account for up to 5% of the total plantation area. Guerin & Abril (2007) measured a methane emission rate of 350 ± 412 kg ha^{-1} yr^{-1} (8.4 ± 9.9 Mg CO_2-eq ha^{-1} yr^{-1}) from a tropical lake in a peat area in French Guiana, suggesting that in the tropics GHG fluxes from open water bodies also have to be considered.

Melling *et al.* (2005b) estimated CH_4 flux from peat soils supporting oil palm, sago and degraded forest, performing monthly measurements over one year using closed chambers. They examined parameters likely to control CH_4 emission: groundwater table, precipitation, nutrients, bulk density, and moisture conditions. The results indicated that the sago plantation and degraded forest were sources for CH_4 while the oil palm plantation was a CH_4 sink. They attributed the switch from the forest as a source (2.27 ug C m^{-2} hr^{-1}) to the oil palm as a sink (-3.58 ug C m^{-2} hr^{-1}) to a lowering of the water table and soil compaction due to use of machinery and concluded that the conversion of tropical peat primary forest to oil palm promoted CH_4 oxidation due to an increased thickness of aerobic soil after drainage. However, increased fire frequency following drainage and management will also increase CH_4 emissions and when vegetation is burned, for each ton of CO_2 emitted, an addional 1.5 kg CH_4 is produced (Scholes *et al.* 1996).

Other CH_4 emission sources

Transformation of forest to agricultural use involves increased management activities such as use of machinery, inputs of fertilizer and mill operations, many of which may promote CH_4 emissions such as those from mill effluent and biomass burning in mill boilers. POME is a major source of methane emission during palm oil production and methods to reduce this are being actively pursued by the industry (RSPO, 2009), which have been estimated estimated at about 32 – 48 kg CH_4 ha^{-1} yr^{-1} (0.8 – 1.2 Mg CO_2-eq ha^{-1} yr^{-1} or 24 – 36 kg C ha^{-1} yr^{-1}) from palm oil mill effluent (Reijnders & Huijbregts (2008).

Nitrous Oxide

Nitrous oxide (N_2O) is primarily emitted as a by-product of nitrification and denitrification in both agricultural landscapes and natural ecosystems. Nitrogen fertilizer use, both inorganic and organic, are a major factor in determining levels of N_2O emission, which vary depending on soil moisture conditions and land use (e.g. Mosier *et al*., 1991; Kroeze *et al*., 1999; Hadi *et al*., 2001; Takadi *et al*., 2006). Natural boreal wetlands with high water tables do not necessarily produce N_2O (Nykanen *et al*., 2002), but may consume small amounts via denitrification when atmospheric N_2O is reduced to N_2. However, tropical peat soils have different biophysical attributes emissions of N_2O from fertilizers and manure may represent addional GHG emissions.

N_2O fluxes have a high temporal variability as shown in a temperate peat in the Netherlands, where three years of half-hourly measurements of N_2O were collected using the eddy covariance methodology (Kroon *et al*., 2010). The large number of measurements allowed the source of N_2O emissions to be differentiated between background emissions and emissions linked to fertilizer application and abrupt climatic events such as rainfall. In this temperate agricultural peat area, N_2O contributed up to 45% to the total GHG balance, when expressed in terms of global warming potential and including CO_2 and CH_4 in the total GHG balance. Event emissions accounted for a considerable part of these N_2O emissions and, therefore, demonstrate the importance to conduct measurements frequently, especially during weather events and fertilizer application.

In oil palm plantations, it seems likely that the application of nitrogen fertilizers will accelerate release of N_2O; however, the extent of those emissions in these types of ecosystems remain poorly documented. Hadi *et al.* (2005) compared the N_2O emissions from a paddy field, a field with a rice-soya bean rotation, and a peat forest (*Table 11*). They integrated monthly measurements and scaled these up to provide annual estimates of N_2O emissions. Takakai *et al.* (2006) estimated an emission of 3.6 – 4.4 Mg CO_2-eq m^{-2} d^{-1} from one year of data by using linear interpolation for temporal upscaling. Melling *et al.* (2007) made monthly measurements of N_2O emissions over one year using closed chambers on tropical peat soils under different vegetation cover: oil palm, sago and forest. In the last study, the N_2O source in the Malaysian oil palm plantations were 1.2 kg N_2O ha^{-1} yr^{-1} (0.48 Mg CO_2-eq ha^{-1} yr^{-1}). However, uncertainties were large and data were too limited either to distinguish background emissions from event emissions due to fertilizer applications and there was too much variability for a robust regression analyses. The default value in the IPCC guidelines for synthetic nitrogen fertilizer-induced emissions for Histosols in tropical regions is 10 kg N_2O-N ha^{-1} yr^{-1} (IPCC, 2006). Based on this value, the N_2O emissions correspond to a total emission of 4.8 Mg CO_2-eq ha^{-1} yr^{-1}. Nitrous oxide emission values for tropical peatlands found in the scientific literature are given in *Table 11*.

Table 11. Nitrous oxide emission values for tropical peat areas as found in the scientific literature, measured by chamber-methodology at different temporal scales.

Reference	Land use on peat	Chamber measurement frequency	Emission (kg CO_2-eq ha^{-1} yr^{-1})
Hadi *et al* (2005)	Rice paddy field	3 measurement days	0-5781
Furukawa *et al* (2005)	Rice paddy field	1 year, monthly	0.016
Hadi *et al* (2005)	Cultivated upland field	3 measurement days	6608-36754
Furukawa *et al* (2005)	Upland cassava field	1 year, monthly	0.257
Melling *et al* (2005)	Sago	10 months, monthly	1556
Hadi *et al* (2005)	Soya	3 measurement days	4543
Hadi *et al* (2005)	Forest, not primary	3 measurement days	6600
Melling *et al* (2005)	Forest, not primary	10 months, monthly	330
Furukawa *et al* (2005)	Forest, not primary	1 year, monthly	0.101
Inubushi *et al* (2003)	Forest, not primary Abandoned upland field rice	1 year, monthly	range -664 - +498
Melling *et al* (2005)	Oil palm	10 months, monthly	566
Furukawa *et al* (2005)	Pineapple	1-2 months	132-1017

Uncertainties and Gaps in Knowledge

In this review, we have attempted to summarize the impacts from the conversion of tropical peatlands into oil palm plantations in terms of both carbon and GHG emissions. All recent pertinent studies have been reviewed and compared; studies differ in the approaches used to assess GHG emissions and there is an element of uncertainty linked to their accuracy and precision.

There has long been a lack of studies that focus on on long-term rates of GHG emissions measured over several years and the uptake of carbon in tropical peats, as well as examining the explanatory variables that mediate the process (e.g. temperature, moisture, chemistry, water table, management, fertilizer inputs). Although recent studies have successfully filled some knowledge gaps, empirical evidence is required to adequately document the relationships between emissions of CO_2, CH_4 and N_2O and their driving variables.

Data on biomass and carbon content in the remnant peat swamp forests are rare and only broad ranges of AGB and emissions rates in peat swamp forests have been documented. On deep peat (>3m) most of the carbon is stored in the peat soil and therefore the relative contribution of the forest carbon stock is less than on shallow peats. Development of a primary (undisturbed) swamp forest into an oil palm plantation will result in a direct release of carbon, ranging between 153 – 200Mg C ha^{-1} due to changes in AGB and peat fire, while development of a logged forest into an oil palm plantation will cause a direct release of carbon, ranging between 47 – 160 C ha^{-1} depending on the degree of forest degradation. The time-avetraged AGB carbon stock of an oil palm plantation is between 24 and 40 t C ha^{-1}, which at the end of each crop cycle is likewise released, or maintained at that amount if a second replanting is pursued.

The conversion of an intact peat swamp to an oil palm plantations releases carbon and GHG to the atmosphere from its AGB and upper peat profiles due to fire. However, these emissions are considered as ‘one - time” emission event. In contrast, the emission linked to drainage and oxidation of peat soils are addional to those initial emissions, and will occur for as long as the soil is drained. Drainage-induced emissions from oil palm plantations on peat have been estimated at about 86 Mg CO_2 ha^{-1} yr^{-1} including the initial emissions peak (Page *et al*., 2011a), with values in the literature ranging from 26 - 146 Mg CO_2 ha^{-1} yr^{-1} (or 7 - 40 Mg C ha^{-1} yr^{-1}) and the most recent estimation is 64 Mg CO_2 ha^{-1} yr-1, with a range between 55-73 Mg CO_2 ha^{-1} yr^{-} for continuous peat emission, excluding the initial peak (Couwenberg & Hooijer , 2013) . Oxidation of drained peat and peat fires are the largest emission sources incurred during oil palm plantation development on peat soils. The processing of FFB and the related production of mill wastes add further to GHG emissions.

The increased fire frequency during clearance and drainage of peat leads to addional in the release of high amounts of CO_2 and CH_4 from both biomass and peat. Based on available measurement data in an abandoned, degraded tropical peat area, the mean burn depth in Indonesia during drought years was estimated at 34 cm per fire event, which translates into approximately 261 Mg C ha^{-1} emission for the years 1997, 2001 and 2002.

Knowledge on CH_4 emissions from tropical peatland is insufficient and only a limited number of short term CH_4 measurements are available. Results are variable and outcomes differ significantly between studies. Based on this very limited number of measurements, terrestrial CH_4 fluxes are estimated to range from 0 - 2 Mg CO_2-eq $ha^{-1}yr^{-1}$ in swamp forests. CH_4 fluxes from open water bodies (drainage ditches and small ponds) have not yet been extensively quantified. Measurment of N_2O emissions in tropical peat systems are likewise scarce and uncertain. The potential N_2O source in an oil palm plantation has been estimated at 566 kg CO_2-eq ha^{-1} yr^{-1} (Melling *et al*., 2007), which is likely to prove conservative. The IPCC (2006) default value for N_2O emissions from fertilized for tropical Histosols is 4.1 Mg CO_2-eq $ha^{-1}yr^{-1}$.

While N_2O and CH_4 should not be ignored, the available data indicates that it is CO_2 that dominates the GHG balance. A point of concern is that in most GHG studies only the 'field' component is taken into account, while emissions from drainage canals, ponds and shallow lakes on subsided or burned land might also be considerable.

Spatial and temporal variations have yet been not fully captured and recent estimates of GHG emissions from tropical peatlands have been based largely on short term studies with high levels of uncertainties due to the reliance on inherently weak methodologies and poor upscaling techniques. Recent studies have started to address these problems, but further field inventories using more technologically sophisticated methods and rigorous experimental design and objective modelling approaches are needed. Because both carbon pools and carbon emissions vary considerably over space and time, the research focus should be on quantification of carbon pools and emissions related to long term land use and land use change at the landscape level.

Carbon release can also take place via waterways (streams, rivers and drainage canals) in the form of dissolved and particulate organic carbon, as well as via dissolved inorganic carbon and CO_2. Studies of these potential carbon flux pathways from tropical peat have been limited, but a recent study suggests that Indonesian rivers, particularly those draining peatland areas, transfer large amounts of DOC into the sea (Moore *et al.* (2013). In that study, it was concluded that the fluvial organic carbon flux from disturbed, drained peat swamp forest is about 50% larger than that of undisturbed peat swamp forest due to land use change and fire.

Recommendations for Reducing GHG emissions

Current sustainability measures in oil palm plantations on peat will decrease the emission source strengths, but will not turn these systems into carbon or GHG sinks. Recent findings suggest that emissions cannot be reduced very much under any management regime when water table depths are around 0.7 m; a common feature of many plantations. Only rehabilitation and restoration of drained peat can turn these systems back into long term carbon sinks.

The simplest measure to limit GHG emissions is to limit or stop development of oil palm plantations on peat. Peat drainage, and thus peat oxidation, and clearance related fires are the largest sources of GHG emissions when establishing oil palm plantations on peat soils. Development of plantations on mineral soil has fewer impacts and impacts are less significant in terms of GHG emissions. If oil palm plantations are developed on peat, oxidation due to drainage will continue either until undrainable levels have been reached, resulting in increased or permanent flooding, or all the peat has disappeared, resulting in exposure of the underlying mineral layers, often potential acid sulphate soils or infertile sands.

The most practical way to reduce GHG emissions in existing plantations is to increase the level of the water table. The RSPO Manual on Best Management Practices for Oil Palm Cultivation on Existing Peat (RSPO, 2012) recommends maintaining water levels in the field at between 40 and 60 cm. If palms are immature, water levels can be as high as 35 to 45 cm below the surface without affecting FFB yield (Mohammed *et al*., 2009). At this level of drainage, GHG emissions can be reduced by more than 50% compared to those with water levels at 70 to 100 cm of depth below the surface. However, flooding should be avoided, because this might enhance methane emissions and reduce FFB yields. To facilitate control of water table depth, correct spacing of drains are required and many exisiting drainage systems need to be modified (RSPO, 2012).

The use of fire for clearing of biomass and the associated burning of drained peat in dry years is the next largest source of GHG emissions in peat swamp areas. The implementation of zero burning and provision of fire prevention measures can help to minimize emissions. Shredding of old palms is a technique that is commonly used to clear old plantations for replanting. The pulverized material can be applied in the field for protection of the soil from drying and erosion and for maintaining soil fertility. Different techniques for pulverization and application of the pulverized materials are examined by Ooi *et al.* (2004). The risk of fire in oil palm plantations on peat is generally reduced when compared to similar peat soil types located in abandoned peatland. Peat and forest fires often occur outside the plantation because of off-site impacts of drainage within the plantations, because the hydrological system surrounding the plantations has been disrupted, which makes these degraded but remnant peat ecosystems susceptible to wildfire.

It is uncertain whether compaction of the peat soil before planting oil palms leads to lower CO_2 emissions compared to no compaction. The oxidation of the peat might be reduced due to the decreased porosity of the soil. Maintenance of a natural vegetation cover of grasses, ferns and mosses and a planted legume cover will reduce decomposition of the peat by reducing soil temperature (Jauhianen *et al*., 2012; Hooijer *et al*., 2012). Maintenance and rehabilitation of hydrological buffer zones can also minimize peat CO_2 emissions from forested areas surrounding plantations (Page *et al*., 2011b).

Recycling of wastes, use of renewable fuels, maximizing fuel savings by using water and rail transport systems, and implementation of mill practices that include CH_4 capture, maximising energy efficiency are possible ways to reduce emissions. The use POME and empty fruit bunches as compost brings addional benefits, as studies show that a40-ton CPO per day capacity mill can provide 20-30% of an estate's fertilizer needs. The use of 'coated' nitrogen fertilizer, composting and careful fertilizer application during rainy seasons will help to reduce N_2O emissions.

Recommendations for Future Research

- Long term measurements are needed of CO_2, CH_4 and N_2O fluxes using a combination of chamber-based measurements to capture small scale spatial variation and eddy covariance measurements to capture temporal variation at the landscape scale. These should be combined with soil subsidence measurements to tackle the very high uncertainties in GHG emission studies.
- Simultaneous recording of variables that may affect the fluxes (e.g. soil temperature, moisture, water table depth, soil and water chemistry, incoming and outgoing radiation) are required to establish robust predictive relationships for GHG models.
- Comparisons should be made of carbon and GHG emissions between ecosystems differing in land use and management intensity (e.g. primary forest, secondary forest, oil palm plantations, and sites varying in depth of water table).
- GHG fluxes of the total ecosystem should be captured, including fluxes from water bodies, using robust, well established, sampling designs.
- In addition to establishing regression models and predictive relationships based on emission data, it is of important to develop methodologies that enable local communities and stakeholders monitor the variables on their holdings that drive the emissions.
- New allometric models should be developed for estimating both above- and below-ground biomass of peat swamp forests and other land cover types prior to establishing plantations (e.g. Verwer & van der Meer, 2010).

OTHER ENVIRONMENTAL IMPACTS OF DEVELOPING OIL PALM PLANTATIONS ON TROPICAL PEAT SWAMPS

With oil palm being the most rapidly expanding crop in Southeast Asia, there is a need to identify sites where the development of oil palm plantations has the least impact, as well as ensure that oil palm that has already been planted enjoys improved management (Wösten *et al*., 2007; Fitzherbert *et al*., 2008). The negative impacts in terms of sustainability of transforming peat swamp forests into oil palm plantations include:

1. Soil subsidence leading to increased flooding risk and salt water intrusion.
2. Loss of biodiversity and ecosystem services.
3. Carbon emissions into the hydrosphere through runoff and erosion.
4. Methane emissions from POME ponds.
5. Discharge of other effluents from palm oil mills into waterways with adverse consequences for water quality.
6. Increased fire risk through peat drainage, leading to adverse implications for human health.

Subsidence, Salt Water Intrusion and Flooding

Tropical peat swamps affect the hydrology of surrounding ecosystems due to their large water storage capacity which slows the passage of flood waters in wet seasons and maintains stream base flows during dry seasons (Yule, 2010). Disruption of this hydrological system, for example by clear cutting and drainage will have consequences for hydrological regulation. For example, because of the low capillary rise in peat soils, oil palm on drained peat is very sensitive to drought and dry periods often result in significant yield reductions (Mantel *et al*., 2007).

Drainage of peat leads to soil subsidence (Polak, 1933; Andriesse, 1988; Dradjad, 2003; Schothorst, 1977; Couwenberg *et al.* 2010; Hooijer *et al*., 2012). Soil subsidence is caused by several processes: consolidation, compaction, oxidation, fires, and water and wind erosion. Consolidation refers to surface height loss caused by tighter packing of the peat soil below the water table. Consolidation of tropical peat drained for plantation development may result in considerable height losses, but usually ends within one year (Den Haan et al., 2012). Like compaction (and shrinkage) of peat above the water table it does not result in carbon losses.

The initial or primary subsidence depends on the type and depth of peat and the drainage level; subsidence rates can be more than 50 cm yr^{-1} in drained tropical peat (Hooijer *et al*., 2012; Wösten *et al*., 1997; Mohammed *et al*., 2009). After a few years of drainage, the balance between the processes contributing to subsidence will change and oxidation becomes the main factor responsible for subsidence. Hooijer *et al.* (2012) indicated that consolidation contributes only about 7% to the total subsidence after the first year after drainage; in fibric peat with low mineral content the role of compaction is reduced rather quickly and becomes negligible after 5 years. Over 18 years of drainage, 92% of the cumulative subsidence was found to be caused by peat oxidation, which is close to the 85-90% reported for subtropical peat by based on more than 76 years of measurements in the Florida Everglades (Stephens *et al.* 1984). Those studies also report that peat surface subsidence continues at a constant rate for many decades, which can explained by the dominance of oxidation and the limited role of compaction (Stephens *et al.* 1984). Wösten *et al*. (1997) report average subsidence rates of 4.6 cm yr^{-1} for oil palm plantations in Johor at 14 to 28 years after drainage (Figure 5). The most recent, extended research of Hooijer *et al*. (2012) shows that constant long-term subsidence rates are 4.5 - 5 cm y^{-1}, on the basis of both literature reviews and subsidence monitoring for water tables between 60 and 80 cm at 218 locations in *Acacia* and oil palm plantations in Indonesia. No studies have been published on the relationship between soil subsidence and CH_4 or N_2O emissions.

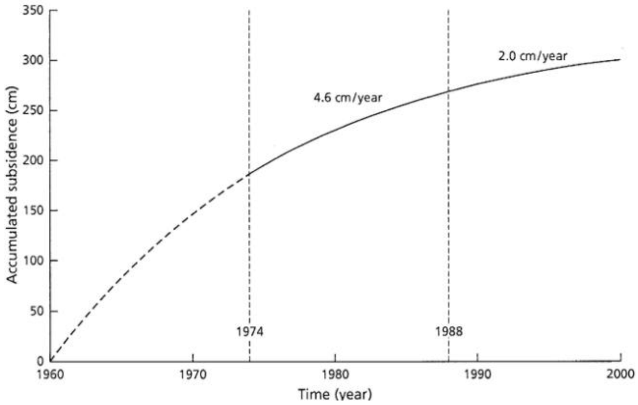

Figure 5. Subsidence rates for individual monitoring locations in relation to depth of water table as measured in *Acacia* plantations six years after drainage, in oil palm plantations 18 years after drainage, and in adjacent forest in Sumatra, Indonesia (Wösten *et al*., 1997).

In the study in Sessang, Sarawak, soil subsidence rates stabilized after 15 years of drainage, ranging from 2.48 cm yr^{-1} in shallow peat (100 – 150 cm), 2.97 cm yr^{-1} in moderately deep peat (150 – 300 cm), and 4.28 cm yr^{-1} in deep peat (> 300 cm). With increasing insight it is more appropriate to split 'first year soil subsidence' from soil subsidence in subsequent years because compaction and consolidation have a greater contribution to soil subsidence in the earlier, than in later years after drainage. In later years subsidence is mainly driven by oxidation.

Soil subsidence can cause the peat surface to drop to levels that enable the water table to reach and rise above the new surface level in periods of high rainfall. This may lead to flooding of adjacent land and downstream areas (Page *et al*., 2009). In addition, because of the soil subsidence and reduced water retention, the freshwater buffer function of the peat swamps decreases, resulting in a decreased buffer against salt water intrusion in the dry seasons (Silvius *et al*., 2000). Examples of the consequences of increased salt water intrusion are, 1) a decline in fish larvae abundance and large scale fish habitats (Cruz *et al*., 2007; Loukos *et al*., 2003), 2) a negative impacts on turtle populations (WWF, 2007), 3) changes in species distribution, reproductive timings, and phenology of ground cover plants (Cruz *et al*., 2007), and 4) impacts on coastal agriculture (Silvius *et al*., 2000). The current sea water rise of about 1-3 mm yr^{-1} in coastal areas of Asia and its projected acceleration to a rate of about 5 mm yr^{-1} over the next century (based on projected climate change with a warming of 0.2 – 0.3 ^{o}C per decade in Indonesia) will amplify the flooding risk (Cruz *et al*., 2007).

With on-going drainage in oil palm plantations the peat will eventually disappear, exposing underlying mineral substrates that will hold far less water and are likely to be nutrient deficient, or, in the case of acid sulphate soils, to contain pyrite (FeS_2) that is detrimental to plant growth (Wösten and Ritzema, 2001). As soon as these soils are drained, pyrite is oxidized and severe acidification results. A number of chemical, biological and physical problems arise from this acidification: aluminium and iron toxicity, decreased availability of phosphate, other nutrient deficiencies, hampered root growth, blockage of drains by ochre, and corrosion of metal and concrete structures. As a result, habitats located downstream of acid sulphate soils may also be threatened (Wösten *et al*., 1997). Exposing these soils will lead to new and difficult problems for local people and land managers (Silvius *et al*., 2000).

To reduce the negative impacts of drainage, such as soil subsidence, high CO_2 emissions, irreversible drying of soils, and eventually drying of oil palm leaves due to moisture stress, the water table has to be managed properly. Mohammed *et al.* (2009) studied soil subsidence in a 1,000 ha peat area in Sarawak, with a peat depth ranging from 100 – 400 cm, and bulk densities ranging from 0.09 g cm^{-3} in deep peat to 0.14g cm^{-3} in shallow peat. The study suggests that sustainably high oil palm yields can be attained by maintaining the water table between -35 and -45 cm from the peat surface after the first two years of planting, with soil subsidence remaining low and CO_2 emissions reduced by 50% compared to more deeply drained soils (Figure 6).

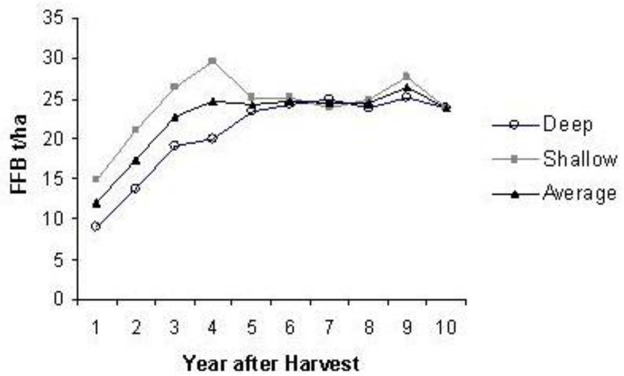

Figure 6. Fresh fruit bunch (FFB) yields of oil palm planted on peat with water table maintained at 35 to 45 cm below field level in the MPOB Research Station in Sessang, Sarawak (Mohammed *et al*., 2009).

Biodiversity

Myers *et al.* (2000) included Malaysia and Indonesia in a list of the top three global biodiversity hotspots. Simbolon & Mirmanto (2000) reported 310 vegetation species in the peat swamp forests of Central Kalimantan. Deforestation and the transformation to oil palm plantations in the tropics has therefore led to a high rate of species decline (e.g. Clements *et al*., 2010; Edwards *et al*., 2010; Wilcove & Koh, 2010; Sodhi *et al*., 2010; Berry *et al*., 2010; Brühl *et al*., 2003; Danielsen *et al*., 2009; Fitzherbert *et al*., 2008; Koh & Wilcove, 2007, 2008, 2009; Hamer *et al.*, 2003). This loss is significant because reductions in species diversity are considered to be irreversible and therefore the need to conserve peat swamp forests in the Indo-Malayan region is clearly urgent (Yule, 2010). Posa *et al.* (2011) have estimated the numbers of species in Southeast Asian peat swamp forests, including those restricted to or strongly associated with this ecosystem (see *Table 13*).

The various types of vegetation on peat all sequester carbon through photosynthesis. Based on the amount of C stored, peat swamp forests are one of the world's most important terrestrial carbon reserves. In terms of usefulness for humans, the diversity of species in the tropical forests is of value for breeding useful animals and plants, as well as for the development of medicines. Among the various types of vegetation in peat swamp forests, some species have high economic value such as Jelutung (*Dyera polyphylla),* whose sap can be used in the production of chewing gum and many other products, and timber species such as Ramin (*Gonystylus bancanus*), Meranti (*Shorea spp.*), Kempas (*Koompassia malaccensis*), Punak (*Tetramerista glabra*), Perepat (*Combretocarpus rotundatus*), Pulai rawa (*Alstonia pneumatophora*), Terentang (*Campnosperma spp.*), Bungur (*Lagastroemia spesiosa*), and Nyatoh (*Palaquium spp.*) (Giesen, 2004). Logging has not adversely affected the fish fauna significantly, but recent incursions such as deepening of drains have increased risks of salt water intrusion (Yule, 2010).

Other than plants, peat swamp forests are the habitat of a number of rare animal species. Tanjung Puting and Sebangau National Parks in Central Kalimantan, both peatland forest ecosystems, are major habitats for the endangered orangutan (*Pongo*) (Gaveau *et al.*, 2009). A number of peat swamp forest areas in Sumatra are habitats for the Sumatran Tiger (*Panthera tigris sumatrana)* and tapir (*Tapirus indicus*). A study by van Eijk & Leeman (2004) in Berbak National Park showed the presence of 107 bird species, 13 mammal species [e.g. wild boar (*Sus scrofa)*, tapir, Sumatran tiger, Malayan sun bear (*Helarctos malayanus*), silvery leaf monkey (*Presbytis cristata*), and Malay stink badger (*Mydaus javenensis*)] and 14 different reptiles and amphibians. Peat swamps in Sumatra, Kalimantan and Papua are also habitats of various endemic fishes, such as arowana (*Scleropages* spp.) (Simbolon, 2011). Sebastian (2002) recorded 57 mammal species and 237 bird species for Malaysian peat swamp forests. Of these, 51% of the mammals and 27% of the bird species were on the IUCN red list of globally threatened species. Regional peat swamp forests are the last refuge for many endangered species from other lowland forests, which are under even greater pressures from logging, hunting and development (e.g. Sodhi *et al*., 2010; Wich *et al*., 2008).

Several authors have proposed strategies that both reduce emissions and enhance biodiversity within oil palm landscapes, such as production of oil palm beneath shade trees, establishment of diverse agro-forestry on plantation boundaries, and maintenance of forest patches within plantations (Koh & Wilcove 2008). A regulation to restrict oil palm expansion to only degraded lands and existing agricultural lands would partly solve the problem. But if logged forests are classified as degraded lands, then biodiversity will continue to decline.

Table 13. Estimated numbers of plant and animal species in peat swamp forests in Southeast Asia (Posa et al., 2011).

Total number of species	Plants	Mammals	Birds	Reptiles	Amphibians	Freshwater fish
Recorded from PSF	1524	123	268	75	27	219
Restricted to PSF	172	0	0	0	0	80
Strongly associated with PSF		6	5	1	3	

PSF, Peat Swamp Forest
Source: Data compiled from various sources available from authors by request

Many of the largest palm oil producers have expressed a desire to implement environmentally friendly management. Maintenance of forest patches within oil palm plantations has been suggested as a means to increase biodiversity. However, Edwards *et al.* (2010) have shown that forest patches, if not inter-connected, did not increase bird abundances in adjacent oil palm, had lower species richness than contiguous forest, and had an avifaunal composition that was more similar to oil palm than to contiguous forest. Another study by Benedick *et al.* (2007) shows that in Borneo, species richness and diversity of butterflies and ants declined significantly with declining forest area and endemic species were not recorded within small forest remnants (<4000 ha). Many studies highlight the importance of retaining areas of contiguous forest for biodiversity protection and they suggest that from a conservation perspective any investment in the retention of forest patches would be better directed toward the protection of contiguous forest (e.g. Berry *et al*., 2010; Edwards *et al*., 2010; Sodhi, 2010; Benedick *et al*., 2007).

The conclusion of Myers *et al.* (2000) is that what we do (or do not do) within the next few decades in terms of biodiversity protection will determine the long-term future of a vital feature of the biosphere, namely the abundance and biodiversity of species. A mixture of regulations, incentives and disincentives targeted at all sectors of the palm oil industry is necessary to protect the region's rapidly disappearing forest (Koh & Wilcove, 2008; 2009). In addition to protecting relatively undisturbed forests, conservation biologists also have to develop strategies to make human-dominated areas more hospitable for forest biodiversity (Gardner *et al*., 2009; Sodhi *et al*., 2010). No conservation strategy can be successful without the cooperation and involvement of local communities. It is, therefore, of great importance to involve local communities and stakeholders in conservation projects, and to create awareness and willingness to cooperate in such schemes.

Emissions to the Hydrosphere

Studies have indicated rising concentrations of dissolved organic caribon (DOC) in past decades in rivers and streams in tropical peat swamp areas. Increases of 15% DOC have been recorded during the transition from dry to wet periods around plantations (Rixen *et al*., 2008). The carbon is transported and rapidly decomposes, causing high fluxes of CO_2 from water bodies (Couwenberg *et al*., 2010; Holden *et al*., 2004). Baum *et al.* (2007) extrapolated DOC losses to the whole of Indonesia and suggested that Indonesia represents some 10% of the global riverine DOC input to the ocean. Rixen *et al.* (2008) suggest that peat soils in the area they studied (the Siak river catchment in central Sumatra) were being destabilized by deforestation, drainage and conversion into oil palm and rubber estates. Anthropogenically enhanced leaching as seen in other studies (Holden, 2005; Holden *et al*., 2004) is very difficult to quantify as base data are usually unavailable prior to deforestation. However, oil palm monocultures are frequently associated with erosion as forest clearance leaves soils bare and exposed to heavy tropical rainstorms before ground cover is re-established. Erosion in turn, causes contamination and sedimentation in water courses. Water quality is also influenced by the runoff of fertilizers into surrounding drainage ditches, causing eutrophic conditions (Rixen *et al*., 2008; Miyamoto *et al*., 2009; Yule & Gomez, 2009). Moore *et al,* (unpublished data) have also shown that deforestation and fire on tropical peat in Central Kalimantan has led to significant increases in fluvial carbon fluxes.

Palm oil processing also has an impact on water quality because palm oil mill effluent (POME) is released into rivers. While the impacts of this are minimised by anaerobic treatment prior to discharge such treatment is predominantly done using open ponds, resulting in large amounts of CH_4 being released into the atmosphere.

Increased Fire Risk

Fires are dependent on four conditions: the presence of fuel (organic material), oxygen, dryness and an ignition factor, and are usually caused by human intervention and linked to activities such as forest clearance, road development, and poor land use management. Undisturbed rainforests usually do not burn, due to high moisture levels in the atmosphere, vegetation and soil. However, drainage, excessive logging and forest clearance disturb the hydrological balance (Langner *et al*., 2007; Page & Rieley, 1998) and make both forests and peat highly susceptible to fires, especially in times of periodically occurring droughts typically coinciding with *El Niño* events (Page *et al*., 2002). Taylor (2010) shows that fire has increasingly affected forests in Indonesia over the last few decades, leading to severe consequences for biodiversity and air quality. Global climate change, coupled with land use changes, could lead to more frequent fires, which in turn could result in positive feedbacks with climate change (Page *et al*., 2002; Hooijer *et al*., 2006; Taylor, 2010). Research suggests that fires were the cause of the largest recorded increase in global CO_2 levels since records began in the 1950s (Aldhous, 2004). The *El Niño* event of 1982-1983 resulted in one of the largest forest fires ever recorded, where four million hectares of forest burnt in Kalimantan and Sabah (Brown, 1998). The fire risk in oil palm plantations on peat is generally reduced compared to that for abandoned, degraded peat land, because of intensive monitoring and control of fires by state agencies and estates (Paramananthan, quoted by Verwer *et al*., 2008).

The consequences of forest and peat fires are numerous and include destruction of the hydrological functioning of peat swamps (e.g. their ability to reduce flood peaks and maintain base flow in periods of drought), a loss of biodiversity and wildlife habitat, the death of seeds and seedlings so preventing re-establishment of vegetation (Yule, 2010), emission of CO_2 and other GHGs (Malhi, 2010), a reduction in photosynthesis due to dense smoke emitted from large fires and thus lower ecosystem production (Hirano *et al*., 2007), and soil erosion.

Another major impact of peat fires with far reaching effects on other ecosystems is air pollution.

Adverse effects on human health in the region have been well documented (Brown, 1998). Forest fires release toxic gases such as carbon monoxide (CO), ozone (O_3) and nitrogen dioxide (NO_2) (Ostermann & Brauer, 2001). At least 20 million people were exposed to dangerously high levels of air pollution during the 1997 fires, with increases in asthma, bronchitis and other respiratory illnesses (Yule, 2010). In addition, many communities rely on forest goods and services such as timber and other forest products as well as water supplies, the quantity and quality of which is dependent on the presence of intact forest.

Discussion and Gaps in Knowledge

Drainage of tropical peat for cultivation leads to soil subsidence that ranges from 2.5 to > 50 cm per year. The subsidence rate is affected by peat type, soil structure, drainage depth and the number of years of drainage. Soil subsidence comprises three processes: compaction, consolidation, and oxidation. Oxidation is the dominant process that drives soil subsidence after the first years of drainage. Soil subsidence can, in the long term, lead to flooding and, in coastal areas to salt water intrusion. Maintaining the water table as high as possible (e.g. 35-60 cm) is the most effective means of reducing soil subsidence. A good practice is to define a 'cut-off' point for cultivation of a plantation before an undrainable level (the drainage base) is reached. This can be defined in terms of a minimum distance between the actual water table and the drainage base.

Tropical peat swamp forests support a rich variety of unique plant and animal species. Transformation of these forests to oil palm plantations always leads to a loss of biodiversity. Many studies highlight the importance of retaining areas of forest and they suggest that the focus should be on protecting existing contiguous forest rather than retention of forest patches within plantations. However, both measures should be encouraged.

Palm oil production on peat is associated with erosion of the drained peat resulting in sedimentation of the waterways and with inputs of fertilizer and crop protection chemicals that act as pollutants. Effluents from palm oil production mills add further to the production and release of wastes leading to further GHG emissions, loss of carbon and adverse effects on aquatic ecosystems.

Peat and forest fires are the second largest GHG sources after emissions due to drainage of peat. Undisturbed peat swamp forests do not usually burn, but can do so if drained and subject to seasonal droughts. Such fires can cause, 1) destruction of the hydrological functioning of the peat swamps, 2) loss of biodiversity and wild life habitats, 3) elimination of seeds and seedlings, 4) release of large amounts of CO_2 and CH_4 to the atmosphere, 5) smoke, resulting in lower ecosystem production, 6) air pollution and adverse effects on human's health, and 7) reduced photosynthesis due in reductions in photosynthetically active radiation (Davies & Unam, 1999a, b)..

Peat fires affect ecosystems worldwide by contributing significantly to climate change through increased GHG emissions. However, information on air pollution associated with the increased fire frequency after peat and forest burning is scarce and more research on these aspects is needed.

SOCIO-ECONOMICS AND PALM OIL PRODUCTION IN SOUTHEAST ASIA'S TROPICAL PEAT LANDS.

Introduction

In the past few decades, palm oil has become a major agricultural product which is used for various purposes such as cooking oil, medicines, pharmaceuticals, animal feed and biodiesel. In general, the raw product, harvested in the form of FFB, passes through various stages before it reaches the consumer. It provides income for many people along this production chain (Kamphuis *et al*., 2011). The oil palm industry is thus part of an economic network ranging from oil palm growers to downstream processing industries (Figure 7). Relations between the different stakeholders are predominantly of an economic and financial nature. The major increase in palm oil production in Indonesia and Malaysia is mainly driven by the global demand for crude palm oil (Kamphuis *et al*., 2011).

Indonesia

The development of oil palm plantations in Indonesia has increased from less than 1 Mha around 1990 to more than 8.1 Mha in recent years (IPOC, 2013). According to Sheil *et al.* (2009) the total planted area in 2009 was 7.3 Mha, of which 5.06 Mha was mature and producing fruit. Indonesian Ministry of Forestry statistics indicate that 70% of the current oil palm estates are located in areas formerly designated as forest for conversion, including over-logged forest (IPOC, 2013; Sheil *et al*., 2009). The large-scale development of plantations in Indonesia is facilitated by different levels of government. An important development in this respect has been the decentralisation of power, which has given local level authorities the right to decide on the use of state land. Large areas of peat forests have been awarded as concessions to private companies and this has resulted in the felling of valuable tree species even in the absence of actual oil palm plantation establishment (Schrevel, 2008). In 2007 the total planted area accounted for over 6.8 Mha of which around 3.4 Mha was controlled by private companies, around 2.8 Mha by smallholders and around 0.7 Mha by public companies.

Malaysia

Plantation development commenced in Peninsular Malaysia at the end of the 19th century (Colchester, 2007a). By 1925, nearly one Mha of land had been cleared of forest and planted with rubber (Jomo *et al*., 2004). Oil palm planting followed and the area of oil palm plantations is still growing, especially in the states of Sabah and Sarawak. In Peninsular Malaysia plantations covered over 2.36 Mha in 2007 (Kamphuis *et al*., 2011). In Malaysia as in Indonesia, there are different sectors involved in the production of palm oil. (2007a) described the example of Sarawak where successive governments since independence in 1963, have supported plantation schemes to promote 'development' and the more productive use of land. Many of the early schemes were with rubber and cocoa. The first pilot scheme with oil palm was implemented in 1966. The crops and techniques may differ but the underlying policy has remained essentially the same while the State has experimented with a series of initiatives to acquire land and capitalize estates in various different ways. None of the schemes have been without problems. Plans continue to promote development of oil palm plantations in so called 'unproductive forest' and in peat swamp forest (Colchester *et al*., 2007a).

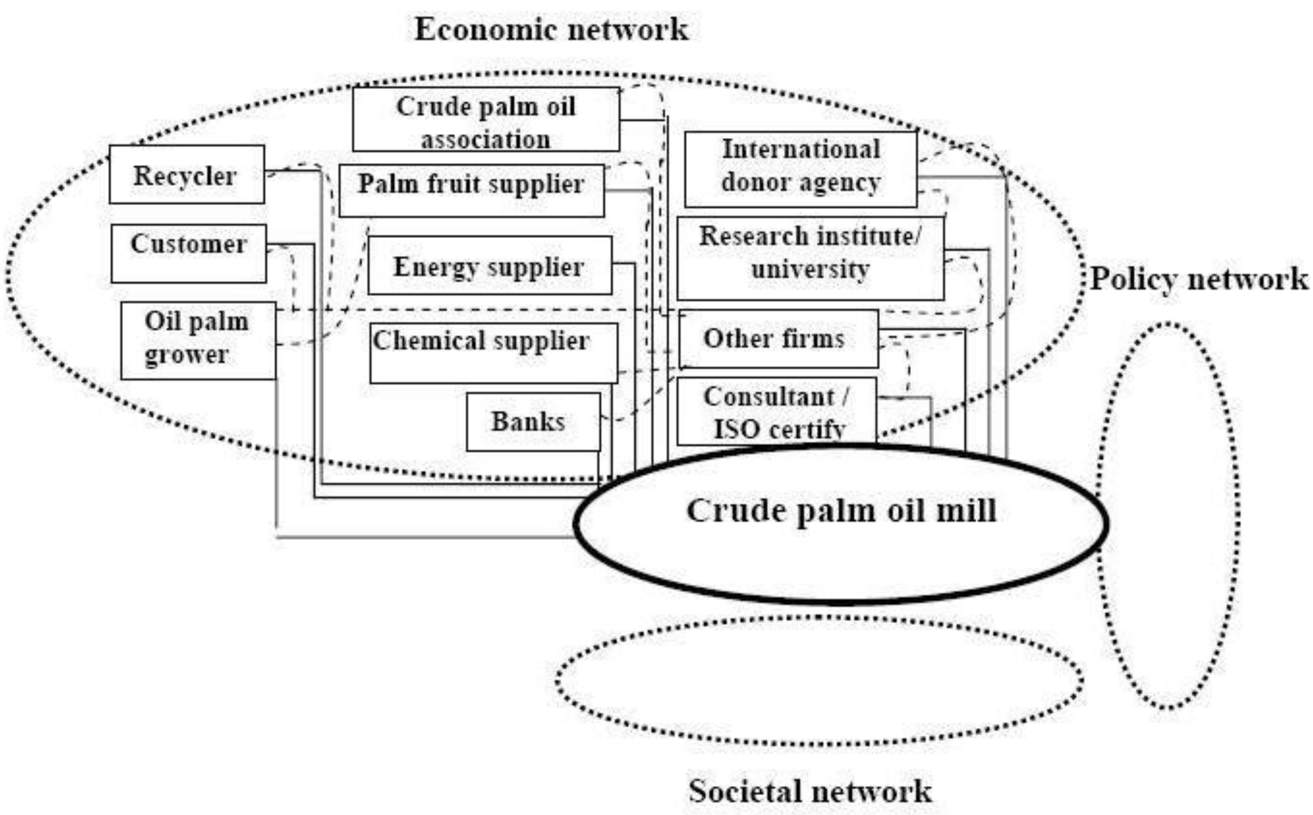

Figure 7. Economic networks relevant to the palm oil industry (adapted from Chavalparit, 2006).

Socioeconomics

Large scale conversion of crops, grasslands, natural and semi-natural ecosystems have social and ecological consequences. Development of estates has often led to negative impacts on ecosystem services and pressure on the remaining natural environment. Some authors have indicated that changes may be irreversible and socio-economic impacts largely negative for the local populations(Schrevel, 2008). The overall economic implications of oil palm as an alternative land use for smallholder income are not yet clear. They differ between regions and type of plantation (Kamphuis *et al*., 2009). Few studies have been published on the economic and social consequences of the transformation of forest to oil palm plantations. Often, studies provide contradictory results and the broader social and livelihood implications of oil palm cultivation remain poorly understood (Rist *et al*., 2009). Some of the reasons that research on this topic is complicated include the large number of stakeholders involved, the interrelationships between actors with different interests, and geographical differences.

Ecosystem Services

Ecosystem Services are the economic benefits that ecosystems provide to humanity (Naidoo *et al*., 2009; Sodhi *et al*., 2010). Tropical forests provide a large number of ecosystem services both at the global level (e.g. climate control) and at the local level, including cultural, provisioning, and regulating services (e.g. erosion control, hydrological control, delivery of natural forest products, fisheries and tourism) (Sodhi *et al*., 2010). Their loss has consequences such as increased erosion, reduced biodiversity, decreases in crop pollination and increased chemical run off, as well as the ecological, social and economic costs of increased fire frequency (Sodhi *et al*., 2010). Also, the large number of people who depend on forest products for their livelihood will be affected by such on-going development.

Forest Dependent Communities

There are serious concerns about the impacts of oil palm expansion on forest dependant communities. Many people who live in rural areas depend on forests for a wide range of goods and services (Wakker, 2005). Conversion of forest has an impact on the livelihoods and culture of these indigenous populations. When forests are replaced by oil palm monocultures, communities lose their access to timber for construction, to rattan and to jungle rubber gardens (Sheil *et al*., 2009), and if they plant oil palm they may become affected by fluctuations in oil palm prices. Many of Indonesia's indigenous people practice shifting cultivation and companies generally prefer hiring workers with backgrounds in sedentary agriculture. For this reason there is a tendency for companies to hire migrant workers, which can lead to ethnic conflict between newcomers and indigenous groups.

Colchester (2007b) interviewed indigenous people in Sarawak and most of them were outspoken in their opposition to the way oil palm plantations are being developed on their lands. They feel their customary rights are being ignored and were promised benefits that were not delivered and measures to secure their consent to proposed schemes to be insincere.

Health

Human health in Southeast Asia has been affected by the haze resulting from ongoing forest and peat fires. Transboundary haze mainly from peat fires has been identified as the most important environmental problem in the ASEAN region. Smoke from tropical fires causes respiratory problems (Kamphuis *et al*., 2011) ,as well as other long-term health problems. Thousands of people died from smoke-related illnesses resulting from forest fires in Indonesia and Brazil (Cochrane, 2003). Components of smoke haze include known carcinogens whose effects may not be apparent for some time. During the 1997 fires, patient visits in Kuching, Sarawak, increased between two and three times and respiratory disease outpatient visits to Kuala Lumpur General Hospital increased from 250 to 800 per day. Effects were found to be greatest for children, the elderly, and people with pre-existing respiratory problems (Sastry, 2000). In Indonesia up to 500,000 people sought hospital treatment for smoke-related illnesses. Health effects depend on the concentration, composition and length of exposure to smoke. The complex mix of particles, liquids and gaseous compounds released depend upon the type and efficiency of burning. These emissions have been studied and quantified for savannah fires but not for tropical forest fires. In addition to respiratory illnesses, blockage of sunlight may promote the spread of harmful bacteria and viruses that would otherwise be killed by ultra-violet B radiation (Beardsley, 1997). Although not all fires leading to smoke haze are set by oil palm plantations and many plantations have adopted zero-burning strategies, there are still well documented cases of large-scale burning by plantation companies and recent analyses by the RSPO GHG Working Group 2 have determined that fires were used in land clearing prior to establishment of many oil palm plantations on peat in recent years.

Employment

Indonesia

The Indonesian oil palm sector has created around three million jobs, the numbers of which are still increasing. Over the next 10 years the Indonesian government plans to double the annual production of palm oil, creating new jobs for an estimated 1.3 million households and reducing poverty for around five million people (Bahroeny, 2009). This has been achieved largely through Nucleus Estate and Smallholders schemes (NES). In these schemes farmers transfer a proportion of their land to an oil palm company for establishment of an estate plantation; the remaining land also being planted by the company but retained as individual smallholdings by the farmers (Rist *et al*., 2010). In some cases smallholders sell their land directly or after one or two years to the company and are paid compensation for loss of land use opportunities. Deals differ significantly in detail, such as in the amount of land given up to the company in relation to that received back as an oil palm smallholding, the amount of debt that the farmer must pay back for the planting of oil palm on the area of land retained, and in the time period over which this must be done (Chong *et al*., 2008; Rist *et al*., 2010).

In 2010 smallholders had a land area of 3.08 Mha, with a share of 35% of the total crude palm oil produced and of 41% of the productive area (Sheil *et al*., 2009; Vermeulen & Goad, 2006). Because of the required machinery and the need for palm oil mills, most smallholder plantations are part of larger, company owned plantations termed nucleus estates (Sheil *et al*., 2009; Kamphuis *et al*., 2011). Wakker (2006) argued that the majority of the economic benefits of oil palm plantations accrue nationally or regionally to a few large palm oil plantation owners and the Indonesian government rather than to smallholders. In addition, because companies prefer experienced labour, large-scale oil palm projects in Indonesia have tended to employ workers from outside the area of operation, fostering social conflicts (Wakker, 2005, 2006; Schrevel, 2008; Wilcove & Koh, 2010; McCarthy & Cramb, 2009). However, these effects are mitigated by the construction of infrastructure and provision of houses, health and educational services that usually accompany the large-scale development of oil palm plantations (Bertule & Twiggs, 2009). As a result, rural communities have easier access to local markets, schools and hospitals.

Malaysia

Oil palm is one of the main drivers of the Malaysian agricultural industry. Malaysia's palm oil industry is the fourth largest contributor to the national economy. Oil palm plantation development started about 100 years ago and production now accounts for 71% of the national agricultural land bank. Malaysia has some of the highest FFB yields at about 21 tonnes ha^{-1} year^{-1}. Malaysia's palm oil industry is regulated by the Malaysian Palm Oil Board (MPOB), which develops policies, guidelines and practices for the industry. As of 2009, Malaysia had 4.7 million hectares of oil palm plantations. The industry is dominated by large plantation companies (both private and government-linked) which hold 60 percent of total plantation land. However, there is a significant proportion of palm oil plantations under the ownership of both organized and independent smallholders who account for 28 and 12% of the total area respectively (Government of Malaysia, 2011). Malaysia's oil palm industry employs a large labour force; MPOB estimated its total size in 2010 in the plantations to be 446 368. This number consists mainly of foreign workers (69%) with locals comprising only 31% (Ramli, 2011).

Income

In Indonesia plantations, particularly oil palm and forestry sectors, contributed 3% to the national economy in 2007 (BAPPENAS, 2009), while the oil palm plantation sector was estimated to contribute 0.85% to GDP. Kessler *et al.* (2007) showed that at a regional level there was a rise in GDP in both the expanding and established regions. At the farm level, the support of the government's nucleus estates that is provided to individual smallholdings has resulted in an increase in income of more than half a million farmers (Zen *et al.*, 2006). The average income for these farmers is seven times higher than the average income of subsistence farmers (Sheil *et al*., 2009). Noormahayu *et al.* (2009)

concluded from a questionnaire study that most of the 200 farmers they interviewed in Sungai Panjang, Malaysia, worked 1.1 - 1.5 ha of land giving an annual average income of RM 5,001 - RM 10,000.

One of the main constraints to such farming was found to be the limited area of land that individual farmers own, which means that most of them plant just one crop, which has no yield during the first 3 years after planting. This renders them vulnerable to exploitation by buyers and other outsiders. Nonetheless, many choose oil palm because it provides a better income than fruit and vegetables. Rist *et al.* (2010) examined the economic implications of oil palm as an alternative land use for smallholders using research sites in Central Sumatra, West Kalimantan, East Kalimantan and Central Kalimantan (see Box 4). They concluded that many smallholders have benefited substantially from the higher returns on land and labour afforded by oil palm, which is in line with published results of Wilcove & Koh (2010), but district authorities, smallholder cooperatives, and the terms under which smallholders engage with palm oil companies, play key roles in the realization of benefits (McCarthy, 2010). Susila (2004) concluded that there is a positive effect on farmers' income generated by palm oil production which reduces income inequality and poverty in palm oil communities. However, income is just one aspect of a sustainable livelihood. The conclusion of Rist *et al.* (2010) is that in Indonesia smallholders are not impoverished by oil palm development but they can suffer by the sale of their land during development. Although Rist *et al.* (2010) show that the cultivation of oil palm may afford new income opportunities to many Indonesian farmers in the short term, they note that the longer term economic implications remain uncertain. Concerns have been raised on topics such as, 1) the adoption of oil palm by smallholders at the expense of more diverse agro-forestry and swidden systems, 2) their vulnerability to crop failure and over dependence on support by companies, and 3) exposure to future economic risk because of price fluctuations or negative ecological impacts (e.g. soil subsidence, exposure of toxic sediments, etc.; Butler *et al*., 2009; Syafriel, 2009; Rist *et al*., 2010; Sheil *et al*., 2009; Schott, 2009).

Box 4

Profile of Smallholders in Siak district, Riau Province, Indonesia.

A group of smallholders are seeking to improve the management of plantations on peat. These smallholders are located in Siak district in two sub districts, Bunga Raya and Pusako, and are organized into seven separate cooperatives coordinated by the *Kelompok Tani* farmers cooperative. With a total membership is about 1,140 families, about 850 families with a total of about 2,200 hectares have elected to pursue RSPO certification with the assistance of the local NGO, Yayasan ELANG.

The total land area is about 3,500 hecatres, all of which is located in shallow peat soils located close by the Siak River. According to PTPN5, a state owned plantation company that collaborates with the smallholders, about 30% of the area has mineral soils and 70% is classified as shallow peat. The plantation was developed under the auspices of the local government with the objective of reducing poverty in the Siak area and to provide opportunities to smallholders for participating in the oil palm supply chain. The project was initiated in 2003 when smallholders were provided assistance to establish oil palm plantations. The establishment was contracted by the local government via PTPN5, which built the drainage ditches and obtained seeds sourced from a reliable seed supplier. The transfer of the plantation from PTPN5 to the smallholders was done in 2009, when the palms first started producing fruit. Assessments of the communities by the RSPO PLWG in 2011 revealed that although many Best Management Practices (BMPs) have been followed, most smallholders were using fertilizer regimes that were better suited for mineral soils and had not yet installed adequate control structures in the drainage ditches in order to maintain appropriate water levels throughout the year. The visit revealed that significant improvements in yield could be made if assistance on implementing BMPs was provided to communities, which would likewise reduce GHG emissions.

Smallholders are sometimes unaware of their rights and the nature of agreements made with companies (Rist *et al*., 2010). Newer, more equitable practices recommended include: 1) the need to clarify smallholder land rights to avoid land tenure conflicts (Chong, 2008), 2) the reformation and standardization of contracts for agreements between farmers and oil palm companies at districts level (Rist *et al*., 2010), 3) the need to improve management capacity of smallholders' cooperatives (in particular, that of the head of the district who plays a key role in raising awareness of rights), and 4) promotion by governments at the national and district level of further oil palm development via individual smallholdings rather than by large businesses (Rist *et al*., 2010). Noormahayu *et al*, (2009) conclude that oil palm cultivation on peat can be a profitable investment so long as growth conditions, costs, selling price and interest rates do not fluctuate substantially.

Summary

The palm oil sector has created millions of jobs and the number of which are still increasing. Oil palm is one of the main drivers of the Malaysian and Indonesian agricultural industry. Oil palm plantation development started about 100 years ago and production now accounts for 71% of the Malaysian agricultural land bank. The Indonesian oil palm sector has created around three million jobs, which are still increasing. Over the next 10 years the Indonesian government plans to double the annual production of palm oil, creating new jobs for an estimated 1.3 million households and reducing poverty for around five million people.

Many smallholders have benefited substantially from the higher returns on land and labour afforded by oil palm. However, in Indonesia, a large part of the economic benefits of oil palm accrue nationally or regionally to relatively few large palm oil companies as well through taxes and fees to the government. Smallholder cooperatives and the terms under which smallholders engage with oil palm companies play key roles in the realization of benefits to local communities. Although the cultivation of oil palm may afford new income opportunities to many local farmers in the short term; the longer term economic implications remain uncertain. Concerns have been raised on topics such as: 1) the adoption of oil palm by smallholders at the expense of, for example, diverse agro-forestry and swidden systems, 2) the vulnerability of smallholders to crop failure and their dependence on companies, and 3) the exposure to future economic risk because of price fluctuations and negative ecological consequences.

Transformation of tropical peat forests to plantations will lead to loss of ecosystem services and affect the social and cultural basis of forest dependant communities. Also health in Southeast Asia has been affected negatively by haze resulting from ongoing burning of above-ground biomass and peat. Health effects depend on the concentration, composition and length of exposure to smoke and include respiratory and cardiovascular complaints among other illnesses.

Knowledge Gaps and Uncertainties

- Information on the social and economic effects of oil palm development is scarce and contradictory.
- There is a major need for alternative production scenarios that allow ecologically and socially sustainable oil palm development and give the highest yields with the lowest social and environmental impacts.
- There is a major need for social studies at all levels, including plantation owners, people depending on forest products or other crops, smallholder cooperatives, and indigenous communities.

MAIN CONCLUSIONS

About 60% of the world's tropical peats are located in Southeast Asia. The original tropical peat swamp forests are important for carbon storage, biodiversity conservation, climate regulation and as a source of for the livelihoods of local communities. The large-scale conversion and drainage of peat swamp forests in Indonesia and Malaysia, in a large part for oil palm plantation development, has significant impacts on the environment.

Currently, most studies indicate that the transformation of an intact peat swamp area to oil palm plantations leads to a release of GHGs to the atmosphere (de Vries *et al.*, 2010; Henson, 2009; Jeanicke *et al.*, 2008; Danielson *et al.*, 2008; Fargioni *et al*., 2008; Rieley *et al.*, 2008; Gibbs *et al.*, 2008; Wösten & Ritzema, 2001; Hooijer *et al.*, 2006). When oil palm plantations are developed on peat, oxidation due to drainage, fires and carbon losses when vegetation is cleared, are major sources of GHG emissions.

Once a plantation is developed on peat, this can lead to serious land degradation over the long term, increased flooding and salt water intrusion into coastal watertables. These conditions also will adversely affect palm oil production eventually.

Effective water management directed at maintaining the water table as high as possible while still maintaining oil palm yield can reduce soil subsidence, GHG emissions and fire risk. Because in all cases peat loss and soil subsidence will continue as long as these landscapes are subject to drainage, a 'cut-off-point' for growing oil palm is recommended before an undrainable level is reached and flooding becomes inevitable.

Methane emission from open water bodies such as drainage canals and ponds is likely to affect the GHG balance. This may be significant as the water surface of drainage canals may account for 2-5% of the total area of a plantation on peat. Better quantification of this emission is required.

Nitrous oxide is primarily emitted from agricultural landscapes as a by-product of nitrification and denitrification. In oil palm plantations the application of N fertilizers and N-containing organic mulches accelerates its release.

The Indonesian and Malaysian oil palm sectors have created millions of jobs and average incomes have risen since oil palm cultivation started. However, although many smallholders have benefited substantially, the majority of the economic benefits accrue to relatively few palm oil companies and to governments. Cooperatives and the terms under which smallholders operate play key roles in the realization of benefits at the local level.

Good implementation of Best Management Practices (RSPO, 2012) in the cultivation of oil palm on peat is necessary to enhance sustainability. However, it is important to note that current sustainability measures in oil palm plantations on peat may decrease emission source strengths, but will not turn these systems into carbon or GHG sinks.

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METHODS FOR DETERMINING GREENHOUSE GAS EMISSIONS AND CARBON STOCKS FROM OIL PALM PLANTATIONS AND THEIR SURROUNDINGS IN TROPICAL PEATLANDS

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ABSTRACT

Tropical peat swamp forests are large carbon reservoirs. These ecosystems store carbon in both above and below ground biomass, but the largest amount of carbon is stored in the underlying peat soil. If a peat ecosystem is converted to agriculture, the carbon stocks will change because of peat drainage and removal of vegetation leading to large greenhouse gas (GHG) emissions. It is important to document and monitor changes in the carbon pools of these globally significant carbon reservoirs, because the GHG emissions that originate from them have the potential to exacerbate global warming. This report discusses available options for measuring, reporting and monitoring carbon stocks and GHG emissions in oil palm plantations and adjacent tropical peatlands.

If an oil palm plantation is developed on peat, the drainage that is required for the cultivation of palms result in continuous emissions of carbon dioxide (CO2), because of the degradation and oxidation of peat. Methane (CH4) and nitrous oxide (N2O) are of less importance in terms of volumes and impacts, but a complete picture requires that all three major anthropogenic greenhouse gases should be considered. Of primary importance is the need to document changes in carbon stock in peat soils that result from plantation management and new plantation developments. In cases where forest is converted, the amount of carbon lost from above ground biomass also needs to be assessed. GHG emissions from soils due to the oxidation of peat can be directly measured using techniques such as the closed chamber method or the more technologically complex eddy covariance method. These direct measurement techniques are used for determining CO2, CH4 and N2O emissions and can be employed to model gas flux over large areas and over time. Indirect methods determine CO2 emission based on measurements of proxies such as soil subsidence and water table depth that link with CO2 emissions.

The methods described in this document have been tested for tracking CO2 emissions in tropical peat ecosystems. For CH4 and N2O, methods have not yet become widely available; consequently, the use of IPCC (International Panel on Climate Change) default values is recommended. Remote sensing technology can be used to monitor changes in land cover, which is a key data input for models that estimate CO2 emissions based on changes in above-ground carbon biomass. Carbon stock values for specific land cover categories are based on allometric equations, while below-ground biomass can be obtained from studies that document root-shoot ratios. Procedures are provided in the annexes for: 1) measuring the water table both in the peat and in ditches, 2) documenting rates of soil subsidence, 3) determining carbon content, bulk density and depth of peat formations, 4) calculating the amount of above- and below-ground biomass, and 5) using destructive sampling of biomass to develop and validate allometric biomass models.

***Keywords**: GHG inventory, oil palm, tropical peat, carbon balance*

INTRODUCTION

The objective of the Roundtable on Sustainable Palm Oil (RSPO) is to promote the growth and use of sustainable oil palm products through credible global standards and engagement of stakeholders. As part of the RSPO, the Peatland Working Group (PLWG) was envisaged as a short-term expert panel established to conduct specific tasks on issues related to the use of tropical peat for palm oil production. One of the objectives of the PLWG was to summarize options for measuring, reporting and verifying greenhouse gas (GHG) emissions and carbon stocks related to the conversion of tropical forest into plantations and from the oil palm plantations established or operating on peatlands.

This document describes the currently used methodologies and is intended to be a user friendly tool to support stakeholders in tracking their carbon gains and GHG emission reductions. Emission reductions can be achieved either through avoidance of emissions by not draining peat for cultivation and avoiding clearance of forest, or through better practices on existing plantations on peat such as improved water management to decrease drainage depth and aid fire control. Measures that can be taken to decrease GHG losses have been described in more detail in the RSPO Best Management Practices Guide and in the RSPO Scientific Review on the Impacts of Development of Oil Palm Plantations on Peat (RSPO, 2012).

The first part of this document provides background information on GHGs and the carbon pools in tropical peat swamp forests. The second part deals with a selection of methodologies that are available currently for measuring and estimating GHG fluxes. Part three covers a selection of methods for determining carbon stocks and carbon stock changes. In addition, we provide information for measuring variables that are needed to calculate carbon stocks and GHG emissions (see Annexes).

Importance of Peatlands

Peatland ecosystems contain a large amount of carbon. They cover approximately 3% of the global land mass and contain 550 Gt of carbon in their soils (note: a gigaton is 10^9 metric tons and is expressed as Pg which 10^{15} grams). Tropical peat soils are estimated to contain around 90 Gt of carbon, with 70 Gt of this being located in Southeast Asia (Page *et al*, 2011a; Jauhiainen *et al*., 2011). The peat soils in Southeast Asia contribute considerably to the global terrestrial carbon stock not only through their underlying peat soil, but also through their above ground biomass (AGB). Indonesian peat soils currently store 50-60 Gt of carbon (132 Gt of CO_2 equivalent) (Page *et al*., 2011a; Jeanicke *et al*., 2008), with above ground stores of about 4.2 Gt (15 Gt CO_2eq.). By comparison, the world's largest rainforest, the Amazon, is estimated to store about 46 Gt of carbon (168 Gt of CO_2eq.) above ground (ICCC, 2012).

The main carbon stocks in tropical peat swamp forest ecosystems are conceptually divided into the following, listed in decreasing mass:

- Soil carbon
- Above ground living biomass (AGB)
- Below ground living biomass (BGB)
- Necromass or wood debris
- Litter

Global awareness of the important role of tropical peatlands in terms of carbon storage has increased, but much uncertainty still exists on its role as a long term carbon pool (Malhi, 2010). As a result of economic development over the past decades, peat swamp forests have been subject to intensive logging, drainage, fires and conversion to plantation estates (Rieley *et al*., 1996, Rieley and Page, 2002). Half of the peat swamp forests in Southeast Asia have been cleared and drained for agricultural use (Hooijer *et al*., 2010), besides, a main part has been degraded through both timber extraction and drainage. The peat drainage that is needed for growing crops such as oil palms causes GHG emissions because of : 1) oxidation of the drained peat, 2) an increase in fire frequency that leads to extensive GHG emissions from the peat and forest cover, and 3) carbon loss during biomass decay even when forest is converted without fire for agricultural development. Conversion of forest represents a one-point emission in time, while emission resulting from peatland drainage is a continuous process.

Greenhouse Gas Emissions

The growth of human population together with industrialization has led to rapid increases in biomass burning, agricultural activities and land use change, resulting in enhanced emissions of aerosols and GHGs into the atmosphere. Changes in the biogeochemical cycles of terrestrial ecosystems, such as the carbon and nitrogen cycles, and their influence on the dynamics of the atmosphere, affect the climate in terms of temperature and precipitation, resulting in increases in droughts, extreme rainfall events and in shifting seasons.

Greenhouse gases reduce heat loss from the Earth's surface, and thus changes in their atmosphere concentrations have a strong impact on climate. Without GHGs, scientists estimate that the average surface temperature on Earth would be approximately 30° Celsius cooler. The key greenhouse gases subject to change in the atmosphere due to human activities are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Their relative impacts expressed as global warming potential (GWP) relative to CO_2 over a 100-year time frame are 25 for CH_4 and 298x for N_2O (IPCC 2006, 2007), meaning that CH_4 and N_2O are both stronger greenhouse gases compared to CO_2 (IPCC, 2007).

Carbon dioxide

Carbon dioxide (CO_2) fluxes between the atmosphere and ecosystems are primarily controlled by photosynthesis and the respiration of plants, animals and soil organisms that mediate decomposition. The balance between photosynthesis and decomposition of organic compounds determines whether a system is a sink (and thus sequesters carbon), or a source (a net emitter of carbon, mainly as CO_2; e.g. Valentini *et al*., 2000). Ecosystems generally act as sources of CO_2 during the night when only respiration occurs and as sinks for CO_2 during the daytime when photosynthesis exceeds respiration. In natural peat soils, the decomposition of organic material is slow because shallow water tables prevent O_2 from penetrating deeply into the soil. Consequently, the degradation of the peat is slower than its rate of production and it therefore accumulates (Alm, 1997). Areas of peat occur worldwide and act as sinks for CO_2 in temperate, boreal and tropical zones. When the peat areas are drained, however, the situation changes: Oxygen penetrates into the soil, the carbon in the peat will oxidize, and CO_2 will be released to the atmosphere.

For tropical peat soils, Couwenberg and Hooijer (2013) concluded that a continuous loss of 55-73 MgCO_2 ha^{-1} yr^{-1} occurs with normal management practice in an oil palm plantation. An initial higher emission rate found immediately after the initial drainage, however, is excluded from this value. The average value of 64 Mg CO_2 ha^{-1} yr^{-1} is in line with the Hooijer *et al*. (2010) and Couwenberg *et al*. (2010) equations, which suggest a CO_2 emission value of ~ 9 Mg CO_2 ha^{-1} yr^{-1} for each 10 cm of drainage depth. The relation between water table and emissions can be used for the purpose of monitoring and reducing emissions within plantations. Changes in CO_2 emission result from land use change, management practices such as drainage (and therefore the oxidation of peat) and from peat and forest fires. CO_2 emissions following drainage of peat soils are the main CO_2 source and are dependent on temperature as well as water level (e.g. Hirano *et al*., 2007; Melling *et al*., 2005b; Couwenberg *et al*., 2010; Furukawa *et al*., 2005; Hooijer *et al*., 2011). Fires may cause even larger CO_2 emissions than drainage but these are generally restricted to relatively short periods of time (Page *et al*., 2002; Couwenberg *et al*., 2010).

Methane

Methane (CH_4) is emitted to the atmosphere as a net result of its production, consumption and transport through soil and water. It is produced in both soil and water bodies, via a microbiological process that occurs when organic matter is degraded in an oxygen starved environment when other terminal electron acceptors (NO_3^-, FE_3^+, SO_4^-) have been depleted by the microbial community (Zehnder & Sturm, 1988). Methane is also consumed in oxygen rich environments, which can vary depending on the residence time of CH_4 within an environment, the oxygen status of the transport route, and the biological activity of that environment. Wetlands, including peat lands, are the largest single source of atmospheric CH_4, and water bodies are also large emitters (Bastviken *et al*., 2004; Walter *et al*., 2006; St. Louis *et al*., 2000; Schrier-Uijl *et al*., 2010c). Methane emissions from wetlands show large spatial and temporal variability. The main factors in temperate regions determining this variability are management, land use history, moisture conditions and climatic conditions, particularly temperature (Moore and Knowles, 1989; Roulet,*et al.,* 1993; Dise, 1993; Segers, 1998; Schrier-Uijl *et al*., 2010a,b,c).

Only a few studies have been conducted on CH_4 fluxes in tropical peat soils (e.g. Ueda *et al*., 2000; Hadi *et al*., 2005; Couwenberg *et al*., 2010; Melling *et al*., 2005a; Furukawa *et al*., 2005). In tropical peat soils, CH_4 emissions usually show a clear positive relationship with water level. However, spatial and temporal variability of the published data are large (Smemo & Yavitt, 2006; Melling *et al*., 2006). Lowering the water table from a depth of 20 cm to a depth of 30 cm led to CH_4 emissions decreasing by 25% (Furukawa, *et al*., 2005). Methane emissions from tropical peatlands are low compared to these in temperate and boreal peatlands, which is most likely due to the recalcitrance of tropical peats (Couwenberg *et al*, 2010).

Nitrous oxide

Nitrous oxide (N_2O) is primarily emitted from agricultural and natural ecosystems as a by-product of nitrification and denitrification (e.g. Hansen *et al*., 1993). Nitrous oxide is naturally present in the Earth's atmosphere, but the atmospheric concentration has increased in recent decades. The increase in N_2O is of concern because it is a long-lived GHG with a global warming potential 298 times that of CO_2 (IPCC, 2007). N_2O is emitted by natural, anthropogenic, and interrelated sources. Natural wetlands with high water tables do not produce significant quantities of N_2O (Nykänen *et al*., 2002) and may even consume small amounts via denitrification, the process where atmospheric N_2O is reduced to N_2 (Regina *et al*., 1996). However, agricultural soils are significant sources of N_2O as a by-product of fertilizer applications (Mosier, 1991; Kroeze *et al*., 1999), and direct N_2O emissions from agricultural soils contribute considerably to the GHG balance (e.g. Kroon *et al*., 2010). N_2O fluxes also have a high spatial and temporal variability, and are difficult to predict (Denmead, *et al*.,1979; Groffmann *et al*., 2000; Velthof *et al*., 1996).

Only a few studies on N_2O fluxes on tropical peat soils under different management regimes are available (e.g. Melling *et al*., 2007; Furukawa *et al*., 2005; Hadi *et al*., 2005). In tropical peat soils, application of nitrogen fertilizers in cultivated systems can accelerate the release of N_2O. Fungi may also play an important role in N_2O production by tropical peat soils (Yanai *et al*., 2007). The extent of N_2O release from the system and the processes that cause N_2O emissions in tropical peat land ecosystems are poorly understood.

Carbon Stocks

Above ground biomass

Above ground biomass (AGB) comprises all the living above-ground vegetation, including stems, branches, twigs and leaves (Verwer & van der Meer, 2010). In tropical forests and in oil palm plantations, the quantity of AGB varies considerably depending on climate, soil type, forest age, forest type, type of undergrowth, etc. Undisturbed primary forests that have the highest biomass are increasingly rare, while the AGB in remnant forests varies with disturbance history and ecological conditions. For example, in peat domes, six forest types have been distinguished (Anderson, 1961), which reflect differences in the hydrology, chemistry and organic matter content of the dome (Page *et al*., 1999). AGB estimates range from 111-432 Mg C ha^{-1} in undisturbed peat swamp forest and from 73-245 t C ha^{-1} in disturbed, logged forest (Waldes & Page, 2002; Agus *et al*., 2009; Lasco, 2002; Gibbs *et al.* 2008; Verwer & van der Meer, 2010). Allometric models based on the correlation between tree height and biomass may be used to estimate carbon stocks in AGB of peat forest.

Below ground biomass

Below ground biomass (BGB) includes all the living organisms in the soil, but is mainly composed of the biomass of living roots (Verwer & van der Meer, 2010). In peatlands it varies from 26.5 Mg ha^{-1} for mixed swamp forest to 14.4 Mg ha^{-1} for low biomass 'pole' forest. In peat swamp forests, roots may be the most important contributors to peat formation (Chimner & Ewel, 2005; Joosten, 2008), but data on root biomass is limited due to difficulties in directly measuring BGB in the challenging conditions of swamp forest (Cairns *et al*., 1997; Jackson *et al*., 1996). For other ecosystems the use of root-shoot ratios has often been used to estimate BGB.

Necromass or wood debris

Necromass or dead wood debris includes all non-living woody biomass not contained in the litter. It may be either standing, lying on the ground, or submerged in the soil. Dead wood includes dead roots and stumps and can be a large component of the total C stock in a tropical peat swamp (Verwer & van der Meer, 2010; UNFCCC, 2010). Research has shown that in tropical moist forests the contribution to the C stock can range between 17 and 58 Mg C ha^{-1} (e.g. Clark *et al*., 2002; Chambers *et al*., 2004; Pyle *et al*., 2008; Palace *et al*., 2008 (in Verwer & van der Meer, 2010), equalling 9.5% - 33.5% of the live AGB (Verwer & van der Meer, 2010). In these studies the carbon content of the wood varied little being between 46% in fully decomposed material to 48% in undecomposed material. A study in Bornean *Dipterocarp* forests revealed an average total necromass of 12.4 t ha^{-1} as standing material and 27.2 Mg C as fallen dead trees (Gale, 2000). Values of 61 Mg C ha^{-1} have been recorded in Borneo for coarse wood debris

(Bruenig, 1996). To measure the biomass and carbon content of the necromass is not easy, especially in mixed stands; it requires considerable labour and it is difficult to obtain an accurate measurement. In oil palm plantations the contribution of necromass is relatively small.

Litter

Litter includes all non-living biomass that is not included in the necromass class. It is the carbon pool that comprises dead organic matter in various states of decomposition at the soil surface, including small dead stems, leaves, small branches, flowers, fruits and seeds (Verwer & van der Meer, 2010; Sayer *et al*., 2007; UNFCCC, 2010). Quantitative data for tropical peat litter is scarce. Because the rate of decomposition in tropical peat soils is slow, the litter biomass pool may be relatively large compared to tropical forest on mineral soils. Nonetheless, the pool is small when compared to the peat itself and hence it does not significantly contribute to total carbon stocks. Estimates of litter carbon pools range from 2.4 Mg C ha^{-1} (Denlaney *et al*., 1997) to 15 Mg C ha^{-1} (Chiti *et al*., 2010). In carbon accounting methodologies the litter pool is often not taken into account because it makes only a minor contribution to total carbon stock.

Soil organic matter

Accumulation of soil organic matter (SOM) originates from remains of plants, including roots, dead leaves, twigs, branches, flowers and fruits. Accumulation occurs because the rate of decomposition of organic matter in wet or water-logged conditions is lower than the rate of deposition. Vast tropical peat deposits containing wood are the result of long-term ecosystem carbon sequestration from the atmosphere. In Indonesia alone, the estimated current peat carbon store is 50 – 60 Pg (Page *et al*., 2011a; Jeanicke *et al*., 2008), with carbon density values between 1500 and 2000 Mg ha^{-1} (Anshari, 2010). Some peatlands, even in natural conditions, are in a steady-state and no longer accumulate peat. Drainage leads to a continuous loss of carbon because of the oxidation and decomposition of peat. The thickness of the peat layer will decrease over time and, with ongoing drainage, the peat soil will subside until the drainage base is reached. Once this occurs the soil may become unsuitable for agriculture due to flooding and intrusion of salt water. The balance between rates of net carbon sequestration by photosynthesis and carbon release by respiration determines whether soil carbon stocks decrease, are in equilibrium, or increase. It has been shown that a high leaf area index and high water levels are essential for net carbon sequestration in peat swamp forest ecosystems (Suzuki 1999; Hirano *et al*., 2007).

Estimating GHG Emissions

Integrated carbon balance

Estimates of above and below ground carbon stocks can be used to estimate changes in carbon balance over time as a consequence of land use or land use change. This requires the determination of the average carbon stock density before and after a certain activity or over a certain period, which typically requires the establishment of permanent sampling plots on different forest classes and non-permanent sampling plots on non-forest land use classes. The number of plots and their location is best determined in a stratified sampling design [Voluntary Carbon Standard (VCS), VM0011, 2012], which involves the following steps:

- Identification of the land use classes and (forest) strata for which carbon stocks are to be quantified;
- A review of existing biomass and biomass increment data for comparison with field measurements;
- Determination of the sample size per land use class or forest stratum;
- Calculation of fluxes from each land use transition category.

The total carbon stock in tropical peat swamp forest consists of carbon stored in AGB and BGB, as well as the carbon stored in the peat. Inputs into the ecosystem include the carbon sequestered by photosynthesis (termed Net Primary Production) resulting in an increase in biomass, and the increase in the accumulated peat. The outputs are the emissions to the atmosphere as a result of autotrophic respiration (cellular respiration in living plants that offsets photosynthesis), and heterotrophic respiration (decomposition), vegetation/crop removal, loss by fire and transport out of the ecosystem by water. Destructive sampling that involves felling, drying and weighing all components of the living biomass, is the most precise method for quantifying biomass within a unit area, but it is labour intensive and hence expensive. Currently, biomass monitoring increasingly depends on *in situ* inventory information supplemented by remote sensing data. In the field, allometric equations, which are regression formulas that relate simple traits such as stem diameter to biomass, are commonly used to estimate AGB. The carbon stored in peat can be determined from measurements of peat depth, bulk density and carbon content.

Integrated greenhouse gas balance

The total GHG balance of an ecosystem consists of 1) all 'natural' sinks and sources of GHG, including fluxes from land and water bodies, and 2) sinks and sources related to agricultural management, such as fertiliser application. The total GHG balance is quantified using the global warming potential of each gas of concern:

$$NEE_{GHG} = NEE_{CO2} + 25NEE_{CH4} + 298NEE_{N2O} \qquad [1]$$

where NEE is the net ecosystem exchange (Mg ha^{-1}) and 25 and 298 are the global warming potentials of CH_4 and N_2O respectively for a 100-year time horizon.

Changes in balances

If biomass is removed or peat is drained the carbon stocks will change and carbon will be lost. Drainage allows oxygen to penetrate deeper into the soil profile, causing aerobic respiration rates to increase with a subsequent release of CO_2. Conversely, if drainage depth is reduced, then less CO_2 will be emitted; however, carbon stocks will still decline by an amount depending on the drainage depth. Because methanogenesis is the result of anaerobic respiration, the emission of CH_4 decreases when water tables fall. Nitrous oxide emissions, which are mainly dependent on the availability of nitrogen, will decrease if management becomes less intensive and less N is applied. Land use changes and activities such as restoration of drained peat lands, changes in water table depth, fires, deforestation, and oil palm plantation development, will all cause a change in the total carbon and GHG balance. To document these changes, carbon stocks and GHG emissions need to be determined before and after the change in each component of the system, and for ongoing emissions (such as these resulting from peat drainage), emissions have to be assessed and documented on a regular basis. One method is to directly measure all carbon and GHG fluxes in the studied ecosystem.

Examples of direct measurement methods for GHG emissions include: 1) discontinuous measurements using small scale enclosures, referred to as the closed chamber method, and 2) continuous measurements at a landscape scale using eddy covariance technology (see below). Examples of indirect approaches are the soil subsidence method and models that use water table depth as a proxy for CO_2 emissions.

For CH_4 and N_2O no proxies are available yet to estimate emissions from tropical peat soil; however, plans are underway to publish revised IPCC Guidelines for National Greenhouse Gas Inventories including a wetlands supplement. These documents will provide revised default emission values for pristine, drained, and rewetted tropical peat for CO_2, CH_4 and N_2O. For above and below ground biomass, carbon stock allometric equations are often used based on inputs such as tree height, stem diameter and root:shoot ratios. If no data are available, IPCC default values can be used.

System Boundaries and Sampling Design

Monitoring carbon and GHG fluxes from an ecosystem involves taking measurements over time and over space. To be effective, the monitoring strategy should:

- Define the 'project area', 'research area' or 'plantation area' in terms of latitude and longitude, delineate peat from non-peat and record any changes that occur during the monitoring period.
- When establishing new plantations, describe land use before plantation development and any land use changes or management activities during the monitoring period. Changes in the surrounding areas should also be included.
- Include all possible GHG sources and sinks within the ecosystem being monitored and include a range of relevant situations (e.g. management activities, meteorological events, etc.) that may affect soil C changes, CO_2, CH_4 and N_2O emissions.
- Stratify the area based on hydrology, land use, management, and vegetation type (see Figure 1) that reflect factors that may control emissions and carbon fluxes. For oil palm plantations on peat strata should be based on differences in ground water depth and the ages of oil palm blocks. Water bodies should be stratified separately.

- For estimating AGB often a 'nested' sampling approach is followed, assessing large diameter trees (with a stem diameter > 30 cm) in rectangular plots of 20 x 2000 m, smaller trees (stem diameter 5-30 cm) in subplots of 5 x 40 m (200 m^2) within these, and understorey vegetation and litter in smaller sub-subplots. Plot location should be randomized if there are marked discontinuities in the vegetation.
- Include information on possible explanatory variables (such as water table, soil moisture conditions, soil and air temperature, inputs of fertilizer, stem diameter, age of tree, and peat depth.
- Insert a soil subsidence pole in each peat strata to monitor soil subsidence and hence estimate carbon losses and GHG emissions from these measurements (see Annex B)

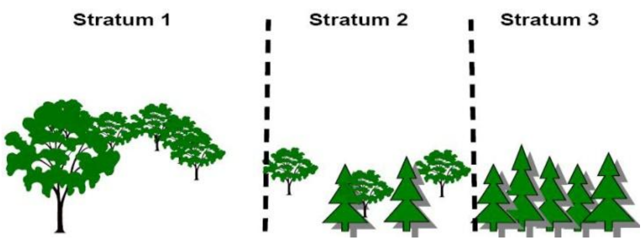

Figure 1. An example of stratification of an area based on forest type.

The spatial sampling design should specify the procedures to be used to summarize data, the number of samples to be taken and the specific sampling pattern. Ideally, a number of land use/management combinations should be defined in each situation and replicates of each combination should be sampled and, if appropriate, stratified by water level classes (Pennock *et al*., 2004). In this document the focus will be on the monitoring of land as opposed to water based fluxes.

In the temporal sampling design based on direct measurements of GHG fluxes, effective data interpolation requires an understanding of temporal patterns of emissions. This temporal behaviour is different for different peat land uses and is dependent on climatic conditions, fertilizer inputs, soil moisture and other conditions that may suddenly change, such as an increase in CH_4 emissions after 'rewetting' of peat. The ability to capture these patterns depends on the temporal spacing and strategy of the measurements.

ESTIMATION OF GHG EMISSIONS FROM PEAT

Emission Inventories

Various measurement techniques and flux estimation methodologies have been developed in recent decades to help create accurate long-term datasets. Methodologies used for emission estimates can in general be classified as 1) direct measurements of GHG and 2) indirect estimates based on variables such as soil subsidence (Table 1). Combining methods and coupling measured emissions with driving variables is useful for up-scaling emissions and for gaining insights into the dynamics of emission fluxes. A sufficiently dense network of observations is necessary to cover the fine scaled spatial patterns that are typical of both managed and degraded peat lands.

Table 1. General comparative characteristics of direct GHG measurement techniques given by Drösler et al., (2008). Capacity/properties range from large/positive (++) to small/negative (- -).

Characteristics	Methods		
	Eddy covariance	Automatic chamber	Manual chamber
Undisturbed gas exchange	++	+/-	+/-
Integration over spatial variability	++	-	-
Direct measurements of small-scale spatial variability and management	- -	+	++
Tracking temporal variability	++	++	-
Costs	- -	- -	++
Workload	++	+	
Performance under all climatic conditions	+/-	+/-	++

On the scale of an oil palm plantation and surrounding forest, the number of chamber-based samples would, depending on the homogeneity of the area and the number of strata, range from between 10 to 20 sample plots, with at least four samples per stratum. Parameters that drive emissions will be different for the three GHGs and will differ for each stratum. Strata can be based on hydrology, land use, management, vegetation type, topography or soil type. If the sampling performed is representative, it can be used to upscale data to larger areas.

Chamber-based methods cannot be performed remotely, while eddy covariance equipment is expensive. For both methods, data processing is time consuming and might only be feasible for selected sites and certain periods, depending on the available technical capacity and budget. Long term datasets are needed to develop, calibrate, and verify, allometric models to arrive at reliable carbon stock estimates. In up-scaling of chamber and Eddy covariance based measurements there are three major challenges: 1) selecting the correct ecosystem variables, 2) developing robust predictive relationships and 3) using long-term datasets (Grofmann *et al*., 2000).

For determining emissions from oil palm plantations, indirect methods are the most feasible for estimating emissions; for example, the use of soil subsidence or water level as proxies. These parameters can predict emissions and are easy to measure in the field; however, they can currently only be used for predicting CO_2 emissions.

Estimating Emissions – The Direct Approach

If data are properly collected and up-scaling is performed in a reliable way, direct GHG measurements are a viable option for determining fluxes from an ecosystem and for understanding the processes that underlie GHG emissions. All possible sources and sinks of CO_2, CH_4 and N_2O should be captured, including photosynthesis, plant- and root respiration, soil respiration, management related fluxes, and losses from water transport.

The closed chamber method (automatic or manual, see Figure 2) (e.g., Melling *et al*., 2005b; Jauhiainen *et al.*, 2012; Schrier-Uijl *et al*., 2010a,b), and the Eddy covariance method (e.g. Hirano *et al*., 2007), have been used successfully for measuring fluxes of CO_2 for many years. However, the physical structure of the ecosystem determines the options available for studies on it. For example, a low or absent vegetation layer, as in the case of fallow agricultural land, facilitates the use of portable chambers (Saarnio *et al*., 2007; Maljanen *et al*., 2007). Closed chambers can also be used to determine respiration at the soil surface below canopies, but the 'tree canopy exchange' is then excluded from the balance. The eddy covariance method, with towers reaching above the top of the canopy, is suitable to capture the total GHG balance of sites with trees, but is limited by its high costs, low portability and low spatial resolution (Lohila *et al*., 2007).

Figure 2. An example of a portable enclosure, connected to a gas analyser and a computer (a). The enclosure is equipped with a water lock to avoid pressure differences and a filter for water vapour to avoid cross-interferences (b). The system can be used to measure CO_2, CH_4 and N_2O on land or on water (a and c) (Photography A. Schrier-Uijl)

Chamber based methods

A closed chamber is placed on the surface to track emissions over time (see Figure 3 for an example). The enclosures function by restricting the volume of available air for exchange across the covered surface, so that any net emissions or uptake of the enclosed gases can be measured as a change in concentration. Closed chambers may be transparent or opaque and to avoid disturbance, are preferably placed on a 'base' that is permanently inserted in the soil at the start of the experiment. The use of transparent chambers allows photosynthesis to proceed by plants located within the enclosure, and thus measures the net ecosystem exchange of CO_2 with the atmosphere. Dark chambers exclude photosynthesis and measure respiration only. Almost all CO_2 measurements from peat in Southeast Asia have been determined using dark chambers, and most studies did not differentiate between autotrophic (root) respiration and heterotrophic respiration arising from the decomposition of peat and litter. Two ways to differentiate between autotrophic and heterotrophic respiration are 1) the use of the so called "trenching approach" (removal of the roots at the chamber location and 2) to measure at different distances from tree positions to cover areas both with and without roots. Jauhiainen and Silvennoinen (2012) combined the trenching approach with measurements made along transects both within and beyond the tree rooting zone in plantations.

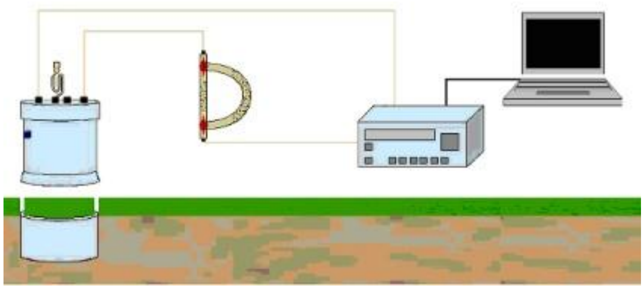

Figure 3. An example of a chamber set-up as used in field experiments. The enclosure for flux chamber measurements, rendered air tight by insertion in the soil, is connected to a gas analyser. The enclosure is equipped with a sensor to correct for temperature, a pressure lock to maintain air pressure inside the chamber, and a moisture filter to prevent interference with water vapour.

The built up of GHGs in the headspace of the measurement chamber is not linear over time (Kroon *et al.*, 2010; Kutzbach *et al.*, 2007), but if measurement times between 4 – 6 minutes are employed, then a linear regression can be used to estimate the slope of the curve (Schrier-Uijl *et al.*, 2010a,b). The use of long enclosure times may underestimate gas flows (e.g. Hutchinson and Mosier, 1981; Pederson *et al.*, 2001) and the use of the intercept-method has been recommended in these cases. The closed chamber method is not recommended for large-scale estimates of GHG emission (Flechard *et al.*, 2007a,b). In most recent studies, the more common defects of using the closed chamber method have been eliminated (Jauhiainen *et al.*, 2012).

Eddy Covariance Methodology

Eddy covariance techniques are used to continuously quantify landscape-scale temporal variability of all three GHG of interest, but are most frequently used to track CO_2 exchanges (Baldocchi, 2003; Aubinet *et al.*, 2000; Veenendaal *et al.*, 2007; Hendriks *et al.*, 2007; Kroon *et al.*, 2007, 2010). The techniques are based on measuring turbulent ascending and descending wind fields, temperature, and gas concentrations at high frequency at specific heights above the soil surface and the vegetation canopy (e.g. Baldocchi, 2003). The eddy covariance method has several advantages: First, it is non-invasive in that it does not change the interactions between soil and atmosphere; second, it integrates gas flows over relatively large areas; and third, it provides continuous measurements over time. The output from the measurements represents the integrated net flux from the landscape upwind from the measurement point. The footprint or area where the signal is coming from is dependent on wind velocity and wind direction and is oval in shape (Figure 4). The extent of the upwind area from which the flux originates depends on atmospheric stability and surface roughness (e.g. Grelle & Lindroth, 1996; Kormann & Meixner, 2001; Neftel *et al.*, 2007). Eddy covariance flux measurements are based on assumptions of horizontal homogeneity, flat terrain and negligible mean vertical wind velocities over the averaging period (Figure 5).

Net ecosystem exchange of CO_2 is determined directly from the flux measurements and is the sum of the gross ecosystem production and ecosystem respiration. The respiration is determined using night time values when photosynthetically active radiation (PAR) is zero and CO_2 flux occurs due solely to respiration. After some corrections, daily averages are estimated from hourly or half hourly data, from which annual balances can be obtained. A regression model can be used to fill data gaps in the CO_2 flux dataset. In temperate regions these models are usually based on temperature (Figure 6). With the currently used instruments, gaps can make up more than 50% of the data set. Gap filling is usually performed by using a regression model based on explanatory variables. Eddy covariance measurements cannot be used to separate autotrophic and heterotrophic respiration because fluxes are measured over an entire ecosystem (e.g. Couwenberg *et al*., 2010; Wetland International, 2009).

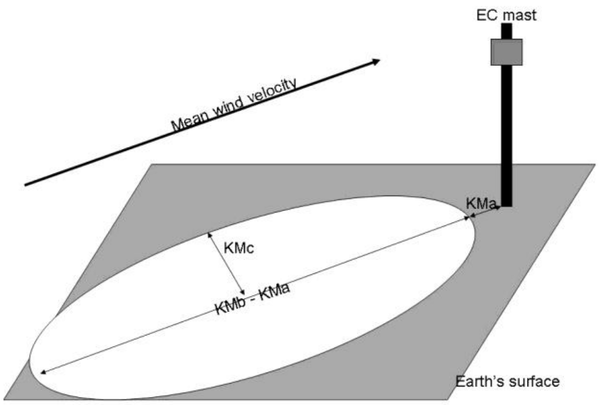

Figure 4. Representation of the footprint area of an EC measurement mast (Kormann & Meixner, 2001). The length of the ellipse (KMb – Kma) and its half width (KMc) are indicated.

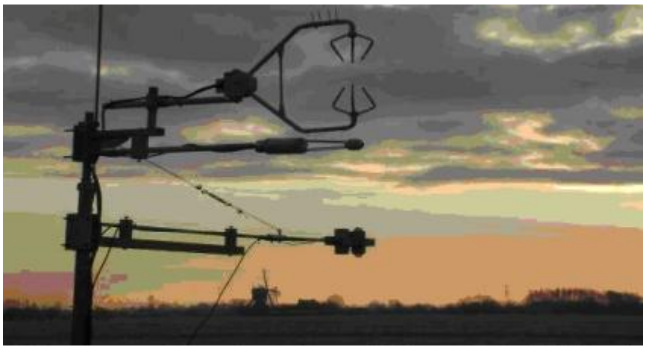

Figure 5. An eddy covariance system, equipped with sensors for irradiation, wind, temperature, CO_2 and water vapour (Photography E. Veenendaal).

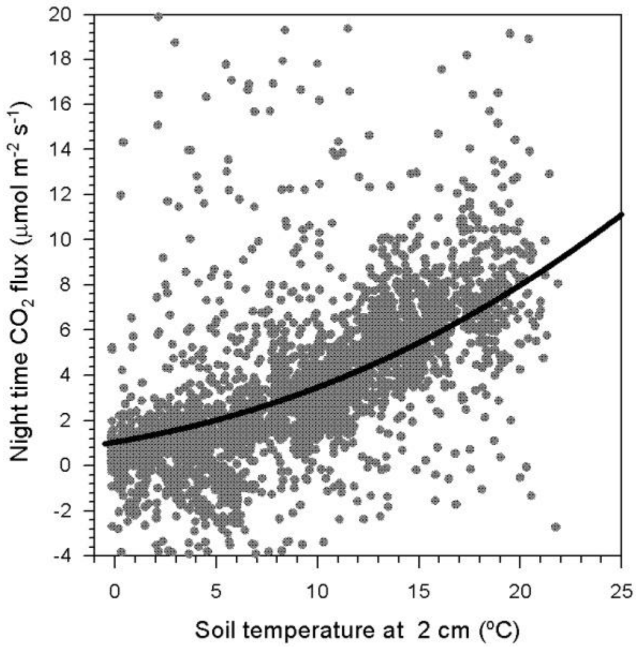

Figure 6. An example of temperature dependency of respiration in temperate regions, adapted from Veenendaal et al., 2007. This model can be used to fill data gaps.

Remote sensing

Remote sensing provides information on land surface properties over large areas, which can be coupled directly to GHG emission factors or to proxies such as water table that are correlated with emission levels. Remote sensing can be used to estimate areas of forest, pasture, open water, croplands and other types of land cover, as well as document land use change over time. It is also a promising method to assess the impacts of fire and logging, which do not always lead to detectable changes in forest cover. Forest degradation due to selective logging is difficult to detect by remote sensing, however. In tropical forests, selective logging usually leaves an intact functional forest, whose canopy will fill in within a year or two and will not appear much different from an unlogged forest with most remote sensing instruments.

Fires contribute substantially to total GHG emissions from tropical peatland, but because of the impact of drought, fires are subject to large inter-annual variation. Fires used in some parts of the world to clear forest for pasture or agriculture, are important sources of GHGs (e.g. Mikaloff Fletcher *et al*., 2004a,b; van der Werf *et al*., 2006; Denman *et al*., 2007). Remote sensing cannot yet be used for estimating carbon losses from peat degradation, although, the area of the fire event and soil subsidence (if areas are bare) can be detected and coupled to CO_2 emissions via the use of empirical models. However, verification on the ground is always needed. A variety of remote sensing methods can be used to identify the location, area, fire scar depth and intensity of fire, and which are able to monitor events responsible for emissions caused by land use or land use change (e.g., peat drainage or deforestation) as well as GHG uptake by sinks (Table 2).

The most effective method for detecting areas of selective logging is to use high spatial and temporal resolution remote sensing data for areas suspected of disturbance. An automated image analysis software using Landsat data and pattern recognition techniques has been successful for detecting selective logging in the Brazilian Amazon (Asner *et al*. 2005). The analysis requires ground-based spectral characterization of surface features and tree species canopy reflectance from a space-borne hyperspectral sensor (Hyperion). The authors found an overall uncertainty of up to 14 percent in the total logged area based on seasonal Landsat data, atmospheric modeling, and detection of forest canopy openings, surface debris, and bare soil exposed by forest disturbances.

Table 2. Current land remote sensing instruments in the public domain.

Instrument	Measurement	Resolution and Coverage	Data Availability
Land Remote Sensing Satellite (Landsat)	Provides the longest continuous record of the Earth's continental surfaces	30-60 m, global	Landsat 7: 1999-present Landsat 5: 1984-present
Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER)	Provides high-resolution images of the land surface, water, ice, and clouds	15-90 m, global	1999-present
Moderate Resolution Imaging Spectrometer (MODIS)	Measures biological and physical processes occurring on the surface of the Earth, in the oceans, and in the lower atmosphere	250 m-1 km, global	1999-present
Airborne Visible/Infrared Imaging Spectrometer (AVIRIS)	Measures constituents of the Earth's surface and atmosphere	5-20 m, aircraft is tasked	1998-present

Modeling Emissions – The Indirect Approach

The indirect approach determines changes in carbon stocks by coupling emissions to one or more variables linked to them. Examples are estimating CO_2 emissions from soil surface subsidence measurements (Hooijer *et al*., 2011; 2012; Couwenberg & Hooijer 2013), or from ground water level measurements (Couwenberg *et al*., 2010; Hooijer *et al*., 2010). These variables are relatively straight forward to measure in the field, but coupling them to emissions has been hampered by a scarcity of long term data. A series of recent studies were designed and executed to overcome this limitation (Hooijer *et al* 2010; 2011; 2012; Couwenberg *et al.*, 2010; Couwenberg & Hooijer (2013). It has to be noted that soil subsidence cannot be used to determine CH_4 or N_2O emissions.

Soil subsidence as a proxy for CO_2 emissions from peat soil

Soil subsidence data provides a time-integrated measurement of the net carbon balance of peat, and is a good alternative to use of direct closed chamber measurements. Using soil subsidence as a proxy for CO_2 emissions integrates changes in soil carbon stocks over a defined time period. Peat subsidence is influenced by various factors:

- Mechanical compaction
- Shrinkage and consolidation after drying
- Decomposition of organic matter
- Leaching of organic carbon via water ways and drainage ditches
- Losses due to fire

Using subsidence avoids the need for instantaneous gas flux measurements, and takes into account most components of the total peat soil carbon budget. Nonetheless, the methodology does have limitations, particularly because CH_4 and N_2O are excluded. Also, while chamber and eddy covariance methods provide reliable annual estimates when measurement protocols span a period of at least one year, the soil subsidence method requires ideally several years of monitoring to estimate annual CO_2 emissions and export by leaching (Page *et al*., 2011a; Hooijer *et al*., 2012). This is partly because annual rates of subsidence are small and the proportion of subsidence due to peat oxidation changes with time. Because of this the intervals for monitoring subsidence need to be more frequent in the years immediately following drainage than in later years. In a stable situation (i.e., 5-10 years after the initiation of drainage), fewer observations are needed compared to the period directly after drainage. Critical to the calculation of CO_2 emissions using soil subsidence is the need to differentiate between consolidation, compaction and oxidation and to adequately validate assumptions regarding the carbon content and bulk densities of the peat formation being evaluated.

Published values of the relative contribution of oxidation of total soil subsidence range from 60% to more than 90% in tropical peat lands. Recent experience shows that it is best to separate the first year's soil subsidence from subsequent years, because immediately following drainage, consolidation plays a larger role in subsidence than in later years (den Haan *et al*., 2012).

In one recent study, Jauhiainen *et al.* (2012) estimated that oxidation contributed about 80% to total subsidence while Hooijer *et al.* (2012) estimated this to be 92%, once rates had stabilized after several years. Lower estimates of the contribution by oxidation to subsidence may be explained by 1) the use of data that apply to the entire period after drainage, including the initial years when compaction and consolidation is indeed dominant, and 2) a reliance on sources describing temperate conditions, where oxidation makes a relatively lower contribution to subsidence due to the biological processes involved being highly temperature dependent (Couwenberg *et al*. 2010; Hooijer *et al*., 2012). In recent research on the relation between emission and soil subsidence in tropical Southeast Asia, only subsidence rate, changes in bulk density, and carbon content of the peat profile were used as inputs to estimate CO_2 emissions (Couwenberg & Hooijer, 2013). Kuikman *et al.* (2003) calculated the CO_2 emissions from temperate peat lands based on soil subsidence as:

$$CO_{2,em} = S_{mv} * \rho_{so} * fr_{ox} * fr_{OS} * fr_C * (44/12) * 10^4 \qquad [2]$$

where,

$CO_{2,em}$ = CO_2 emission (kg CO_2 ha^{-1} yr^{-1})
S_{mv} = soil subsidence rate (m yr^{-1})
ρ_{so} = bulk density of the peat (kg m^{-3})
fr_o = oxidation fraction of the peat (-).
fr_{OS} = fraction organic matter in peat (-) on weight base.
fr_C = carbon fraction in organic matter (-).

The same equation can be used for tropical peatland, with fr_{ox}, fr_{os} , fr_C and ρ_{so}. being derived from available data or from literature.

See the appendices for procedures to determine bulk density of peat, the fraction of organic matter in peat, and the carbon fraction of the organic matter. Guidelines for measuring soil subsidence are given in Annex B.

Water level as a proxy for CO_2 emissions from peat soil

Several studies in Southeast Asia have estimated emissions based on drainage depth. For example, Couwenberg *et al*. (2010) coupled soil subsidence and water level to emissions and calculated emissions of at least 9 Mg CO_2 ha^{-1} yr^{-1} for each 10 cm of drainage depth. Consequently, changing an un-drained peat swamp forest with the water table at the soil surface to a drained peat area with a drainage depth of 60-80 cm would lead to an emission of between 54-72 Mg CO_2 ha^{-1} yr^{-1}. The relation between water table depth and daytime CO_2 emission is also described by Jauhiainen *et al.* (2012) by using a linear regression where autotrophic and heterotrophic respiration have been separated using a 'trenching' approach and allometric equations for daytime oxidation emission at locations furthest from trees (which exclude root respiration as much as possible):

$$CO_2 \text{ emission } (mg\ m^{-2}\ hr^{-1}) = 953.35 \times \text{drainage depth } (m) + 309.07 \qquad [5]$$
$$(R^2 = 0.47, SE = 197)$$

Both the relations found by Couwenberg *et al*. (2010) and Jauhiainen *et al*. (2012) (equation 5) can be used to determine CO_2 emissions from drained peat. Guidelines for measuring the water table depth are given in Annex A.

IPCC default values for CH_4 and N_2O emissions

The CH_4 emissions from tropical wetlands show, in some cases, a possible exponential relationship with water level when this is close to the soil surface (Jungkunst & Fiedler, 2007; Huttunen *et al*., 2003). However, knowledge of CH_4 and N_2O emissions from tropical peat ecosystems and their underlying processes is sparse. Methane emissions in fields where the water level is more than 20 cm below the surface are considered negligible. In cases where no relevant data are available and cannot be gathered (e.g. for N_2O and CH_4), IPCC default values can be used to calculate emissions but IPCC default values are general and have a large uncertainty as they are not specifically developed for a particular site.

ESTIMATION OF CARBON STOCKS

Estimation of carbon stocks in above ground biomass

Different steps need to be taken to estimate the total AGB in a forest, a plantation and other areas. Estimates per biomass stratum are usually obtained from regression models based on different field variables such as diameter of trees at breast height (DBH), tree height, and crown area of individual trees of varying diameters. Tree species can differ between project areas, and on peat, often a stratified pattern of species composition can be found depending on the location in the peat dome (Figure 7). The sample size should be large enough to capture the spatial variability of trees within the project boundary. Two methods for estimating their biomass are discussed here.

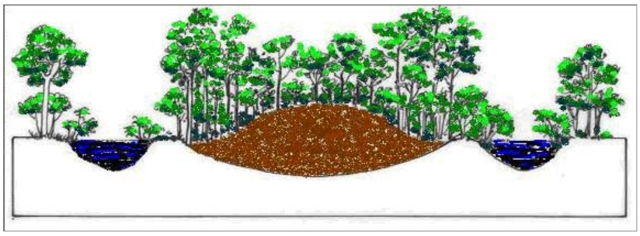

Figure 7. Schematic cross section through a peat dome.

Estimating of tree biomass using the BEF method

One approach for estimating tree biomass and carbon stock is the use of a biomass expansion factor (BEF) and basic wood density for the conversion of stem biomass to AGB for individual tree species. The AGB is than expanded to total tree biomass using the root-shoot ratio. The biomass of individual trees of a particular species (B_{tree}) is estimated as:

$$B_{tree} = V_{tree} \text{ x } WD \text{ x } BEF \text{ x } (1 + R) \qquad [6]$$

where B_{tree} is the dry biomass in Mg, V_{tree} is the stem volume of the tree (m^3), *WD* is the basic wood density (Mg dry matter m^3), *BEF* is the biomass expansion factor for conversion of stem biomass to AGB (dimensionless) and *R* is the root-shoot ratio for the tree species (dimensionless). The biomass expansion factor, root-shoot ratio and basic wood density data for a certain species can be obtained from either: 1) local sources of site specific data; 2) National sources of species-specific data (e.g. national forest inventory or national GHG inventory; 3) Globally available data applicable to the species; or 4) IPCC default values.

Allometric equations for estimating above ground biomass

Allometric models that are available in the scientific literature are developed using destructive sampling of individual trees; to date, not many have been developed for trees in peat swamp forests. In these models, a relationship is constructed between a combination of the stem diameter, tree height and crown area and the biomass of each tree observed. One such formula is:

$$Ln\ (AGB) = -0.744 + 2.188 \text{ x } ln\ (DBH) + 0.832 \text{ x } ln\ (WD) \qquad [7]$$

where *WD* (g cm^{-3}) is wood density expressed as mass per unit volume (g cm$^{-2;}$ see Verwer & van der Meer, 2010 and Basuki *et al*., 2009), and *DBH* is stem diameter at breast height in cm. Allometric relationships between tree diameter and AGB have also been developed by Hashimoto (2004):

$$Ln\ (AGB) = a \text{ x } ln\ (DBH) + b \qquad [8]$$

where *a* and *b* are coefficients which vary depending upon species (see Table 3). Chave *et al* (2005) tested different models and concluded that the best overall model for AGB of wet forest if the tree height is known was:

$$AGB = 0.0776 \text{ x } (\rho\ DBH^2\ H)^{0.940} \qquad [9]$$

and if the tree height is not known, was:

$$AGB = \rho \times \exp\,[-1.239 + 1.980\ ln\,(DBH) + 0.207\ ln\,(DBH)]^2 - 0.0281\ ln\,(DBH)^3 \qquad [10]$$

with AGB in kg, DBH (diameter) in cm, ρ (oven dry wood volume over green volume) in g/cm^3 and H (tree height) in meters.

Guidelines are given in Annex E for estimating AGB, in Annex G, for estimating tree height, and in Annex F for destructive sampling of the under-storey and litter layer. Carbon stock estimations for the non-tree vegetation components are usually based on destructive harvesting, drying and weighting.

Validation of existing allometric models

It is desirable to validate the applicability of allometric equations. In all cases, source data from which equations were derived should be reviewed in order to confirm they are representative of the forest type, species composition and conditions in the research area. Allometric equations can be validated by:

1. Limited Measurements (VCS, VMD0001 vs1, 2011)
    - Select at least 30 trees. The minimum diameter of measured trees should be 20 cm and the maximum diameter should reflect the largest trees present or potentially present in the project area.
    - If validating a forest type-specific equation, selection should be representative of the species composition in the project area, with species being represented roughly in proportion to their relative basal area.
    - Measure DBH, and to the first branch.
    - Calculate stem volume from measurements and multiply by species-specific density to obtain biomass.
    - Apply a coefficient (biomass expansion factor) to estimate total AGB from stem biomass.
    - Plot the estimated biomass of all the measured trees along with the curve of biomass against diameter as predicted by the allometric equation.

      If the estimated volumes of the measured trees are distributed both above and below the curve (as predicted by the allometric equation) the equation may be used.

2. Destructive Sampling (VCS, VMD0001 vs1, 2011; details of destructive sampling are given in Brown, 1997)
    - Select at least five trees representative of the species composition in the project.
    - Measure DBH and height and calculate the volume of the bole or stem.
    - Fell and weigh the AGB to determine the total wet mass of the stem, branch, twig, leaves, etc. This may be done by extracting and immediately weighing sub samples of each of the stem and branch components, followed by oven drying at 700 Celsius to determine dry biomass.
    - Determine the total dry weight of each tree from the averaged ratios of wet and dry weights of the stem and branch components.
    - Plot the estimated biomass of all measured trees to create a curve of biomass against diameter.

Table 3. Statistical details of allometric equations for estimating above ground biomass of some tree species (Hashimoto *et al.*, 2004).

Species	a	b	N samples	Min	Max	Adjusted r^2	Adjusted means of ln (Wt)
Ficus sp.	2.60	-2.59	26	3.5	9.1	0.95	1.55
Geunsia pentandra	2.62	-2.89	20	3.4	16.2	0.91	1.31
Piper aduncum	2.39	-2.42	37	3.2	8.3	0.92	1.38
Other species	2.40	-2.49	108	3.2	20.3	0.81	1.36
All species combined	**2.44**	**-2.51**	**191**	**3.2**	**20.3**	**0.85**	

Estimation of above ground carbon stocks in oil palm plantations

The AGB of oil palms and their related carbon stock is dependent on the age of the tree. Khasanah *et al*. (2012) determined an average value of 34 Mg C ha^{-1} for oil palms on peat and they also derived an empirical relation between palm height and the amount of C stored in the palm. Alternatively, destructive sampling could be used to estimate oil palm carbon stock. Melling *et al* (2008) reported values of 19 Mg C ha^{-1} stored in the stems, canopy and roots of five year old oil palms with an additional 7 Mg C ha^{-1} in the palm's by-products. Once the biomass is estimated, C stocks of each pool within an oil palm plantation can be calculated by multiplying the biomass by the carbon content (Table 4).

Table 4. Average carbon content of the different components of live and dead biomass in oil palm plantations.

Component	C content (fraction)
Live biomass	
Palm biomass	0.47
Understory	0.47
Root	0.47
Dead organic matter	
Litter	0.4
Dead wood	0.5

Estimation of Carbon Stocks in Below Ground Biomass

The BGB pool is smaller than the AGB biomass pool (see Table 5 for examples for peat swamp forest). Roots have often been neglected in studies due to difficulties in measurements and are usually included within the total soil carbon pool. Allometric models are available for estimating the below ground root biomass in peat swamp forest (Niiyama *et al*., 2005; Sierra *et al*., 2007; Kenzo *et al*., 2009; Niiyama *et al*., 2010). An example is the equation of Niiyama *et al* (2010):

$$BGB = 0.02186 \times DBH^{2.487} \quad [11]$$

where *BGB* is below ground biomass and *DBH* = stem diameter at breast height (1.3 m).

Another approach is to estimate BGB from root:shoot ratios described in the literature. Sulistiyanto *et al*., (2007) estimated that the BGB varies from 26.5 Mg ha^{-1} for mixed swamp forest to 14.4 Mg ha^{-1} for low pole forest, giving a root:shoot ratio of 1:12 for mixed swamp forest and 1:18 for low pole forest. For oil palm root biomass a root:shoot ratio of 1:4 has been used (Khasanah *et al*., 2012), but this value is expected to be updated once more data become available. IPCC uses the root:shoot ratios shown in Table 6. The carbon content of the BGB biomass is usually around 50 – 60%.

Table 5. Estimated biomass and carbon content of relatively undisturbed peat swamp forest based on plots measured by Anderson between 1954 and 1960 in Sarawak, Malaysia (Anderson 1961). Note the order of magnitude difference in carbon stock between the different biomass compartments.

	Biomass (Mg /ha) Average ± SD	Carbon content (Mg C/ha) Average ± SD
Above-ground biomass	338.8 ± 52.5	169.4 ± 26.4
Below ground biomass	74.0 ± 15.8	37.0 ± 7.9
Coarse woody debris	62.9 ± 9.8	31.4 ± 4.9
Litter	7.9 ± 1.2	3.9 ± 0.6

Table 6. Root: shoot ratio's used by IPCC 2006 for the calculation of below ground biomass.

Domain	Ecological zone	Above-ground biomass	Root : shoot ratio	Range
Tropical	Rainforest	< 125 Mg/ha	0.20	0.09 – 0.25
		> 125 Mg/ha	0.24	0.22 – 0.33
	Dry Forest	< 20 Mg/ha	0.56	0.28 – 0.68
		> 20 Mg/ha	0.28	0.27 – 0.28
Subtropical	Humid forest	< 125 Mg/ha	0.20	0.09 – 0.25
		> 125 Mg/ha	0.24	0.22 – 0.33
	Dry forest	< 20 Mg/ha	0.56	0.28 – 0.68
		> 20 Mg/ha	0.28	0.27 – 0.28

Estimation of Carbon Stocks in Below Ground Biomass

The range of coarse woody debris (CWD) biomass or necromass in peat swamp forest can be estimated roughly using the ratio between woody debris and AGB found in non-peat tropical forests and applying this ratio to the AGB calculated for peat swamp forests. Verwer & van der Meer (2010) published a table on biomass carbon stock of coarse woody debris based on different studies in tropical moist and wet forests. On average these studies revealed that the course woody debris biomass pool may comprise 9.5 – 33.6 % of the live AGB pool. Verwer & van der Meer (2010) calculated the average minimum and maximum CWD necromass and carbon content of swamp forest, based on historical inventories and an assumed AGB:CWD ratio. However, their estimates are based on studies carried out in non-peat swamp forest. For lying dead wood the VCS approved 'methodology for assessing carbon gains through avoided planned deforestation of un-drained peat swamp forests' can be used.

Estimation of Carbon Stocks in Peat

The carbon storage capacity of peat (C_{peat}) depends on its bulk density, carbon content and depth. Thus:

$$C_{peat}\ (t\ C\text{-}org) = A\ (ha) \times D\ (m) \times BD\ (t\ m^{-3}) \times C\ (\%) \qquad [12]$$

where A is the total area of peat in hectares, D is the average peat depth in meters, BD is bulk density in tonnes per m^3 and C is the percent carbon on a dry weight basis.

Changes in carbon stock are determined by the differences in these parameters over time. Tropical peats are heterogeneous and surface BD can range from 0.05 t/m^3 to 0.25 t/m^3 for a well compacted peat. A procedure that is commonly used to estimate BD in peat soils is given in Annex C. At least two different techniques have been used for determining the BD of peat; the auger method as used in most earlier studies and the 'pit' method used more recently to avoid compaction artefacts associated with augers (Hooijer *et al.* 2012). Plantation development on peat requires compaction before planting to create better rooting conditions for anchoring palm trees and to facilitate access by workers and machinery. The compaction starts after clearing and continues until planting starts, resulting in a higher surface BD compared to that of undisturbed peat soil.

On a dry weight basis, a carbon content of around 55% is representative of hemic and fibric tropical peat in Southeast Asia (Wösten *et al.* (1997); Page *et al.*, 2011a; and Couwenberg *et al.* (2010) reported similar values. Other data on carbon content in tropical peat are provided by Warren *et al.* (2012), Hooijer *et al.* (2012), Dommain *et al.* (2011) and Page *et al.* (2004) in Central Kalimantan, all in the range of 45-60%. If the value for the carbon content is not available, IPCC default values can be used (IPCC, 2006).

Peat depth can be measured by drilling until the mineral soil is reached. Existing peat depth maps can also be used in combination with interpolation techniques to derive more detailed maps. These maps should then be validated by field measurements.

Guidelines for estimating the peat depth and carbon content of the peat are given in Annex D.

Losses through peat fires are calculated by determining the area of the fire scar and the reduction in peat depth caused by the fire, which is then multiplied by the mean bulk density and carbon content to give the amount of carbon lost. Most of the carbon will be lost as CO_2 to the atmosphere.

CONCLUSIONS

There are a variety of methodological approaches that can be used to estimate GHG emissions linked to the establishment and operation of oil palm plantations in tropical peatlands. There are essentially two approaches: 1) directly measure GHG flows, and 2) document changes in total carbon stocks. In both instances measurements must take into account both spatial and temporal variability linked to natural and human disturbance. Sampling intensity and methodological rigour can have a large impact on both accuracy and precision, particularly when field data are plugged into numerical models used to estimate changes in carbon stocks or when scaling up GHG fluxes across soil profiles and vegetation canopies. Perhaps the most practical approach is to develop proxies that can be used to track changes and estimate emissions, rather than attempting to measure GHG emissions or changes in carbon stocks directly. Proxies should be parameters that are easy to collect (e.g., soil subsidence for peat soils and remote sensing for AGB), but they must be validated by detailed studies that establish the relationship between the proxy and the subject being monitored, preferably using regional or local data sets. If no data are available, official data from comparable sites may be use, or global IPCC default values can be used, particularly in the case of CH_4 and N_2O emissions from peat soils and carbon loss from drainage ditches.

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