Estimation of the impact of oil palm plantation establishment on greenhouse gas balance

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Abstract Estimates of emissions indicate that if tropical grassland is rehabilitated by oil palm plantations, carbon fixation in plantation biomass and soil organic matter not only neutralises emissions caused by grassland conversion, but also results in the net removal of about 135 Mg carbon dioxide per hectare from the atmosphere. In contrast, the emission from forest conversion clearly exceeds the potential carbon fixation of oil palm plantings. Forest conversion on mineral soils to promote continued oil palm mono cropping causes a net release of approximately 650 Mg carbon dioxide equivalents per hectare, while the emission from peat forest conversion is even higher due to the decomposition of drained peat and the resulting emission of carbon oxide and nitrous oxide. The conversion of one hectare of forest on peat releases over 1,300 Mg carbon dioxide equivalents during the first 25-year cycle of oil palm growth. Depending on the peat depth, continuous decomposition augments the emission with each additional cycle at a magnitude of 800 Mg carbon dioxide equivalents per hectare.

The creation of 'flexibility mechanisms' such as the clean development mechanism and emission trading in the Kyoto Protocol could incorporate plantations as carbon sinks in the effort to meet emission targets. Thus, for the oil palm industry, grassland rehabilitation is an option to preserve natural forest, avoid emissions and, if the sequestered carbon becomes tradable, an opportunity to generate additional revenue.

Keywords Carbon sequestration · Emission · Green house gas · Land rehabilitation · Savanna · Tropical forest · Kyoto Protocol

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1 Introduction

The contention regarding the environmental impact of the oil palm (*Elaeis guineensis* Jacquin) industry is primarily about the conversion of tropical rain forest into plantations (Wakker, 1999). Once established, oil palm plantations managed in an environmentally friendly way are sustainable production systems (Basiron & Weng, 2004; Ibrahim, 1992). Most of the area needed for the expansion of the oil palm industry was supplied through the conversion of forest. This expansion and the increased productivity were the main factors that enabled the industry to become a globally important provider of affordable vegetable fat. Oil palm is harvested in many tropical countries on more than 12 million hectares and yields over 32 million tonnes of oil annually. It accounts for more than one quarter of the global vegetable oil market and is the most important oil crop next to soybean. During the past 5 years the oil palm area harvested expanded at a rate of 400,000 ha per year (FAOSTAT data, 2005). The frequently discussed employment of palm oil derivates as a substitute for mineral oil products may additionally contribute to the expansion of oil palm cultivation.

Further growth of the oil palm industry may take place on land currently covered by lowland forest, degraded grassland and agricultural land under alternative use. Undulating Oxisols and Ultisols supporting forest vegetation are considered as prime land for oil palm. In Southeast Asia most of these areas are already under agricultural use or declared as nature reserves (Mutert, Fairhurst, & von Uexküll, 1999). Recently, in Borneo and Sumatra, there has been a distinct move towards conversion of forests on peat soil into oil palm plantations. An option to provide land for oil palm planting without threatening the future of tropical rain forests is the rehabilitation of anthropogenic grassland, created by human clearance of natural forest biomes a long time ago. The principal steps in successful rehabilitation are the removal of the vegetation cover that is often dominated by speargrass or cogongrass (*Imperata cylindrica* (L.) Beauv.), liming, phosphorus fertilisation and establishment of a fast growing leguminous cover crop (von Uexküll & Mutert, 1994). In Indonesia alone there exist 20×10^6 ha (von Uexküll & Mutert, 1994) to 50×10^6 ha (Goldammer, 1993) of anthropogenic grassland. In West Africa (Chikoye, Manyong, & Ekeleme, 2000) and in parts of tropical America (Bulla & Lourido, 1980; Houghton, Lefkowitz, & Skole, 1991; Moreira, 2000) degraded land is abundant as well. It has been suggested that in some countries the commitment to protect natural forests and the high profitability of oil palm might lead to an increased replacement of other tree crops such as cacao and rubber. Within a country, such a replacement may occasionally be of importance, but on a global scale it has no significance. In Malaysia, for example, oil palm planting has expanded from $.3 \times 10^6$ to 2.5×10^6 ha since 1974, while the area planted to cacao has decreased by only $.5 \times 10^6$ ha. In Indonesia, Thailand, Papua New Guinea and Nigeria, planting of both cacao and oil palm has expanded. Worldwide, the area under cacao cultivation has expanded by 61% and rubber area by 60% within the last three decades (FAOSTAT data, 2005). Thus, oil palm will not replace rubber or cacao, but may drive them into other areas such as natural forest land. Therefore, most of the future spread of oil palm plantations is likely to take place either on forest land or rehabilitated grassland.

Industrial plantation designers inevitably tend to choose the biologically prolific sites in good climate, with seemingly rich soil, good water supply, and easy access,

rather than the remote, poor quality habitats with damaged soil and little vegetation. Therefore, the new huge industrial plantations will negatively impact or destroy some of the most pristine ecosystems on the earth (Patzek & Pimentel, 2005). The 'flexibility mechanisms' such as the clean development mechanism and emission trading of the Kyoto Protocol could act however as an incentive for grassland rehabilitation.

The environmental impact of oil palm plantation establishment may be categorised in three principal effects: change in the greenhouse gas balance, erosion (Hamer, 1981; Maene, Thong, Ong, & Mokhtaruddin, 1979) and reduction of biodiversity by fragmentation, disturbance and destruction of natural habitats (Laidlaw, 2000; Robertson & Schaik, 2001). This paper focuses on the major fluxes between sources and sinks of greenhouse gases (GHG) during land use change due to oil palm plantation establishment. The aim is to assess the volume of GHGs emitted in the course of oil palm plantation establishment.

2 Methods

Changes of above and below ground biomass (BGB) and soil organic matter and the resulting greenhouse gas fluxes in response to oil palm plantation establishment are assessed. Three land types are considered: forest on mineral soil, forest on organic soil and anthropogenic grassland. A tool for carbon-credit traders, national and international bureaucrats to estimate the magnitude of emissions caused by land use change is provided with the 'Guidelines for National Greenhouse Gas Inventories' by the 'Intergovernmental Panel of Climate Change' (IPCC, 1997). The assessment of greenhouse gas emissions by oil palm establishment is prepared in conformity with these guidelines where possible. Alternative values published in current literature are used where the IPCC fails to provide default values and where provided values appear not to represent the local situation. The averages calculated from literature data are given with the respective standard deviation. In contrast to the aim of the IPCC guidelines, which focuses on annual balances, only the total change in the greenhouse gas balance occurring within the economic life span of oil palm is assessed. The economic life span of oil palm, which is assumed here to be 25 years, is limited in plantations by the increase of harvesting expenditures with the palm's growth in height.

Gases included in the calculations are those referred to in 'Annex A' of the Kyoto Protocol: carbon dioxide, methane and nitrous oxide. The emission of CH_4 and N_2O are converted by multiplication with their global warming potential into CO_2 equivalents, i.e., the quantity of CO_2 that is equally absorbent of solar radiation. According to the IPCC Third Assessment Report, the most recent global warming potential values for a 100-year time horizon are 23 for CH_4 and 296 for nitrous oxide (IPCC, 2001).

3 Emission from decomposition and burning of forest and anthropogenic grassland

3.1 Biomass of forest vegetation

The biomass of tropical rain forest varies greatly in response to the local environment. Case studies in the Malaysian state of Sarawak revealed that above ground

biomass (AGB) of different forest types in the same region varies within a range of 210–650 Mg ha^{-1} (Proctor, Anderson, Chai, & Vallack, 1983). Further, inconsistency in methods applied causes considerable disparity between biomass estimation results. Brown and Lugo (1982, 1984) calculated tropical forest AGB from two distinct data sources: timber volume and destructive sampling. Using volume data, they calculated 176 Mg ha^{-1} AGB of closed primary tropical forest, whereas destructive sampling yielded 283 Mg ha^{-1} AGB.

The biomass of tropical lowland forests, the forest type most frequently converted for oil palm growing, is usually higher than that of upland forest (Brown & Lugo, 1984). According to Whitmore (1990) the AGB of a typical tropical lowland primary forest is about 400 Mg ha^{-1}. However, the AGB actually encountered may be considerably lower, as the values from few forest types do not reflect the spatial heterogeneity of forest areas caused by differences in environmental factors and human impact (Brown, Iverson, Prasad, & Liu, 1993). Logging or fragmentation causes a marked loss of AGB due to a sharp increase in tree mortality and tree damage near fragment margins (Laurance, Laurance, & Delamonica, 1998). Lasco (2002) reviewed published data and derived from these a typical AGB loss through logging from 22% to 67%.

For forests where annual precipitation exceeds 2000 mm, the IPCC quotes AGB default values of 300 Mg ha^{-1} in Africa, 295 Mg ha^{-1} in America, 225 Mg ha^{-1} in continental Asia and 275 Mg ha^{-1} in insular Asia. The corresponding values for moist forests with a short dry season are about 20–50% lower. It has been pointed out, however, that these values are a starting point for emission estimates and that local conditions need to be considered for accurate values (IPCC, 1997). The emission calculations in this study are based on 73 datasets of AGB from lowland forests found in an oil palm suitable climate. Data are taken from Brown and Lugo (1984), Brown (1997), Brown and Gaston (2001), Lasco (2002), Proctor, Anderson, Fogden, & Vallack (1983), Sanford and Cuevas (1996), Vitousek and Sanford (1986) and Whitmore (1984, 1990). The derived mean for AGB of 295 ± 152 Mg ha^{-1} is in the vicinity of the IPCC default values while its rather high standard deviation underlines the necessity to apply local values.

The BGB to AGB ratios of tropical forest available in literature vary widely, too. A review of case studies reveals ratios from as low as .03 to up to .82 (Brown, 1997). A BGB to AGB ratio of .13 for tropical lowland forest is given by the IPCC (1997). Sanford and Cuevas (1996) calculated from literature data a ratio of .14, whereas Brown and Lugo (1984) used narrower ratios to estimate BGB of tropical forests. They applied a ratio of .17 for tropical moist forests and .20 for tropical wet forests. Whitmore (1984) estimated the BGB of tropical forests in the Far East with a ratio of .20 to .25. The mean ratio of all references evaluated in this work is .18 and thus slightly higher than the IPCC default. The average ratio calculated from the references is applied to estimate the BGB where only AGB values were supplied, which is the case for 41 of all 73 tropical forest AGB values considered. Accordingly, the tropical lowland BGB was calculated to be 47 ± 26 Mg ha^{-1}.

Based on the above figures, the total above and BGB of tropical lowland forest in oil palm suitable environment is 342 ± 178 Mg ha^{-1}.

3.2 Biomass of grassland vegetation

The grassland biomass is determined by the floristic composition, precipitation, soil properties, fire, wildlife and other factors (Mistry, 2000) and varies greatly in response to one or more of these. The AGB of grass-dominated savanna on poor sandy soils in Venezuela is as low as 2 Mg ha^{-1} (Bulla & Lourido, 1980), while it reaches over 25 Mg ha^{-1} with growing shrub and tree density (de Castro & Kauffman, 1998; Scholes & Hall, 1996). For grassland in the humid Guinea zone of Africa, Goldammer (1997) states an AGB of up to 8 Mg ha^{-1}, which is close to the IPCC default value for tropical savanna ranging from 4.9 Mg ha^{-1} to 6.6 Mg ha^{-1} (IPCC, 1997). The IPCC guidelines define savannas as vegetation formations with a predominantly continuous grass cover.

A review of literature on typical *Imperata* grassland in oil palm growing regions reveals AGBs ranging from 3.8 Mg ha^{-1} to 23.0 Mg ha^{-1} (Hartemink, 2001; Hashimoto, Kojima, Tange, & Sasaki, 2000; Holmes, Lemerle, & Schottler, 1980; Lasco et al., 2000; NRI, 1996; Otsamo, 2001; Roshetko, Delaney, Hairiah, & Purnomosidhi, 2002; Wibowo, Suharti, Sagala, Hibani, & van Noordwijk, 1997). The average AGB of 11.2 ± 7.3 Mg ha^{-1} obtained from these data and applied in the following calculations is distinctly higher than the given IPCC range. It is hypothesised that this difference in biomass reflects the usually high soil fertility and favourable rainfall in areas suitable for oil palm production. The large standard deviation is in all probability not only a response to environmental conditions but mirrors the continuous loss of soil fertility in fire-controlled grassland. Magcale-Macandog (2002) has shown that the annual loss of AGB in grassland reduces *Imperata* leaf production by 47% within 28 years.

No default value for grassland root biomass is given in the IPCC reference manual and studies published on grassland root biomass in humid tropics are limited. Results published by Lasco et al. (2000) and Chikoye and Ekeleme (2003) provide AGB to BGB ratios of .7 and 1.0. A comprehensive review by Jackson et al. (1996) concludes that the root to shoot ratio of tropical grassland is .7. Other investigations of grassland root biomass noted distinctly greater ratios. BGB was higher than AGB by a factor of 1.5 in East Kalimantan and by 1.3 in Sumatra (Syahrinudin, 2005). Fearnside (2000) calculated from literature data an average root to shoot ratio of 3.1 for central Brazilian scrub savanna. Some researchers found even higher root to shoot ratios of up to 7 (Bulla & Lourido, 1980; de Castro & Kauffman, 1998; Delitti, Pausas, & Burger, 2001). The latter ratios were obtained from measurements of tropical grassland on fast draining soil and in areas of low precipitation. As these locations are unsuitable for large-scale oil palm cultivation, the respective ratios will not be considered further. Applying a mean ratio of 1.4, grassland BGB of 15.5 ± 10.1 Mg ha^{-1} is computed from the AGB values.

The process by which invasive grasses replace forest is closely linked to fire, which kills young trees and seedlings but allows grasses to survive (MacKinnon, Hatta, Halim, & Mangalik, 1996). Annual burning of grassland (Andersson, Michelsen, Jensen, & Kjoller, 2004; Otsamo, 2001; van der Werf, Randerson, Collatz, & Giglio, 2003), a linear development of the AGB and insignificant BGB fluctuation is assumed for the estimations. Based on the above values and assumptions, the time-averaged biomass of grassland (above and below ground) in oil palm suitable environments of the humid tropics is 26.7 ± 17.4 Mg ha^{-1}.

4 Emission from biomass decomposition

Oil palm plantation establishment requires the removal of the existing forest or grassland plant cover. After clearing, the biomass is, if not burned, broken down by litter-feeding invertebrates like termites, earthworms and beetles. By reducing litter to tiny particles, they increase the surface area for bacterial and fungal decomposition (MacKinnon et al., 1996). Decomposition releases plant nutrients that become available again to plants and emits the carbon contained in the biomass into the atmosphere as carbon dioxide (CO_2). A fraction of the carbon is released as methane (CH_4) through the activity of termites. Due to the uncertainty of the effect of clearing on termite populations and associated CH_4 release, no guidance on calculation of this component is included in the IPCC methodology (IPCC, 1997). Hence, the CO_2 release by decomposition is estimated as a direct function of biomass volume and carbon content. For all forest carbon stock estimates, a vegetation-independent carbon content of 50% is assumed (IPCC, 1997), while for *Imperata cylindrica* grassland a carbon content of 43 is applied (Hartemink, 2004; Syahrinudin, 2005).

Biological decomposition of small litter (e.g., leaves and twigs) in the tropics may reach 50% in less than 1-year, whereas decomposition of hardwood tree species takes more time (Anderson et al., 1983). A study in the central Amazon revealed that an average tree decomposes by 95% in 18 years (Chambers, Higuchi, Schimel, Ferreira, & Melack, 2000). After cutting the vegetation there is an initial rapid loss of easily decomposable root biomass, leaving behind a large fraction of resistant material (Gijsman, Alarcón, & Thomas, 1997). Arunachalam, Pandey, Tripathi, and Maithani (1996) found that fine roots (diameter <2 mm) of tropical forest regrowth in humid subtropical climate decompose in 600 days by more than 82%. A study of soil under pasture to a depth of 6 cm showed that the initial decomposition rate is high, but that remains of forest roots are present for at least 7 years (García-Oliva, Casar, Morales, & Maass, 1994). References describing the fate of the resistant root fraction of cleared vegetation deeper in the soil profile are scarce. Further, no recommendations for the estimation of emissions from decomposing root biomass are provided by the IPCC guidelines. Due to the long time frame of 25 years considered in this work, a complete decomposition of all cleared above ground and root biomass is assumed.

Based on the above considerations, the complete decay of 1 Mg cut biomass results in an emission of 1.8 Mg CO_2. This translates into a total emission from biomass decomposition of 42.0 ± 27.4 Mg CO_2 per hectare of grassland and 627.0 ± 326.3 Mg CO_2 per hectare of forest (Table 1).

5 Emission from biomass burning

Burning of the cleared vegetation is a common practice in the conversion of forest into arable land to reduce the bulky biomass and facilitate subsequent fieldwork (Fearnside, 2000). In a growing number of countries regulations have been introduced that restrict or prohibit the use of fire in land clearing. However, land preparation without burning involves higher clearing costs and consequently these regulations are frequently disobeyed. The scale on which such policies are disregarded became obvious during events such as the 1997/1998 El Niño Southern

Table 1 Carbon dioxide emission (Mg ha^{-1}) as a result of complete decomposition of above and below ground biomass of forest or grassland vegetation after converting the land into an oil palm plantation

	AGB	BGB	Total biomass	Carbon content	CO_2 emission
Forest	295 ± 152	47 ± 26	342 ± 178	171 ± 89	627 ± 326
Grassland	11 ± 7	15 ± 10	27 ± 17	11 ± 7	42 ± 27

Oscillation when many plantation companies in Indonesia used the draught to clear land (Casson, 2000). During this time, 176 companies were accused of illegal burning; 133 of these belonged to the oil palm industry (Wakker, 1999).

The biomass of grassland is comparatively low and burning is not necessary to assist in oil palm planting. Furthermore, burning is known to significantly reduce soil carbon (Bird, Veenendaal, Moyo, Lloyd, & Frost, 2000) and may result in increased weed pressure as fire fosters *Imperata cylindrica* dominance (King & Grace, 2000). Nonetheless, fire can control rat and other pest populations and reduce the risk of accidental fire after oil palm planting. Therefore, burning is also considered in the grassland emission assessments.

The combustion rate expresses the proportion of biomass consumed by fire, which varies tremendously over short distances within any given clearing (Fearnside, 2000). The IPCC guidelines state a default combustion fraction of 50% for cleared forest biomass. However, it is recommended that the value be adjusted to actual local conditions (IPCC, 1997). For oil palm plantings, part of the cleared biomass might be stacked in the palm inter-rows to decompose. Thus, a lower combustion fraction than for field crops is acceptable. Efficiency of a simple burn often does not exceed 20% (Araujo, Carvalho, Higuchi, Brasil, & Mesquita, 1999). If less than 30% of AGB is burned, Hartley (1988) suggests pushing the remaining material in windrows and to undertake a second burn. As a result of repeated burning, some 40% of the carbon contained in the AGB from forest clearing enters the atmosphere through combustion, while the rest is released through decomposition (Andreae, 1991). The emission assessment in this study is based on a 40% combustion fraction of cleared forest AGB.

Grassland AGB consists mainly of inflammable material, which admits a higher combustion fraction than in forest clearings. The IPCC recommends general default values in the range from 80% to 85%, if detailed local information is not available (IPCC, 1997). The combustion fraction of AGB in Brazilian bush savanna is reported to range between 84% and 92% (de Castro & Kauffman, 1998). While the lower value was measured for savanna, which contains a higher proportion of woody biomass, the higher value represents the result obtained from mainly grass and herb vegetation. In this study a mean combustion fraction of 85% is applied for the emission assessment of grassland.

Biomass burning immediately releases a part of the total carbon in cleared vegetation, while some of the remainder is converted into charcoal, which is resistant to decay (Fearnside, 2000). Due to the low proportion and the eventual decay (IPCC, 1997), charcoal formation is not considered in the emission calculations.

In the case of grassland rehabilitation, the total emission from AGB burning and the decay of unburned above and BGB is 43.5 ± 28.3 Mg CO_{2eq} per hectare. Combustion plus decomposition of forest biomass would otherwise release 648.0 ± 337.2 Mg CO_{2eq} from each hectare (Table 2).

Table 2 Carbon dioxide equivalent emission (Mg ha^{-1}) caused by burning and decomposition of above and below ground biomass of forest or grassland vegetation during the first 25-year oil palm rotation

	AGB	BGB	AGB carbon burned	CO_{2eq} emission from AGB burned	Carbon in BGB and unburned AGB	CO_2 from decomposition	Total CO_{2eq} emission
Forest	295 ± 152	47 ± 26	59 ± 30	237 ± 122	112 ± 59	411 ± 215	648 ± 337
Grassland	11 ± 7	15 ± 10	4 ± 3	16 ± 11	7 ± 5	27 ± 18	43 ± 28

6 Changes in soil organic carbon

The humid tropics exert a disproportionately large influence over terrestrial C storage, but C allocation and below ground retention in these ecosystems remain poorly quantified (Giardina, Binkley, Ryan, Fownes, & Senock, 2004; Houghton, 2005). Soil C stocks and potential fluxes of GHG are primarily determined by the soil type and, accordingly, the IPCC approach differentiates between mineral and organic soils to estimate emissions.

For mineral soils, the IPCC gives, depending on soil activity, an approximate quantity of 5–180 Mg ha^{-1} of organic C in the uppermost 30 cm under native forest in tropical moist to wet climate (IPCC, 1997). Excluding the IPCC approximations for regions with extended dry periods and sandy soils unsuitable for oil palm cultivation, a range of 60–180 Mg ha^{-1} is obtained. Considering the vertical distribution of soil organic carbon in forest soils, at an average of 50% in the top 20 cm (Jobbagy & Jackson, 2000), this is coherent with a comprehensive study on carbon distribution in tropical Asian forests. For seasonal and moist lowland forests, reviewed data and estimates indicate a soil organic carbon density of 144 Mg C ha^{-1} in the upper 100 cm (Brown et al., 1993).

In the course of forest conversion, a considerable part of soil C is typically lost through decomposition (Houghton & Goodale, 2004). Despite obvious alterations in soil colour and bulk density, which indicate a change in soil organic matter during oil palm establishment on native forest land, scientific investigation and documentation on respective carbon dynamics is scanty. Therefore, the emission calculations will be based on an average of 120 ± 60 Mg C ha^{-1}, representing the IPCC values for the upper soil layer under forest in an oil palm suitable environment. The primary consideration in the IPCC guidelines regarding C stock changes in the topsoil derives from Sombroek, Nachtergaele, and Hebel (1993), who argue that in most soils, management effects on soil carbon at depth are minimal. This has also been found by Fearnside and Barbosa (1998), but they point out that the ultimate contribution from deep layers to emission may be substantial. Despite the potential importance of subsoil C stock changes after deforestation and plantation establishment, these are not integrated here due to the limited knowledge available on the respective dynamics.

The IPCC guidelines suggest the application of the following formula to estimate changes in soil carbon (IPCC, 1997):

$$SC_M = SC_N \times \text{base factor} \times \text{tillage factor} \times \text{input factor}$$

where SC_M is the amount of soil carbon after land use change and SC_N is the amount of soil carbon under native vegetation. The 'base factor', the principal factor accounting for soil carbon changes, is .7 for long-term cultivated land and .5 for degraded land. In no-till systems the 'tillage factor' is 1.1 and the 'input factor' is 1.0 where crop residues are retained. For sites where forest is converted into plantation land, the multiplication of the base and tillage factors gives a value very close to the average carbon loss of 30% found in a review of over 100 different studies, most of them conducted in tropical environments (Murty, Kirschbaum, McMurtrie, & McGilvray, 2002).

Applying the above formula, the conversion of natural forest into an oil palm plantation on mineral soil reduces the initial soil organic carbon stock of 120 ± 60 Mg C ha^{-1} by 40.8 ± 20.4 Mg C ha^{-1}, while the increase of soil organic matter under plantations established on rehabilitated grassland leads to a net fixation of 13.2 ± 6.6 Mg C ha^{-1}. The magnitudes are consistent with findings from Mutuo, Cadisch, Albrecht, Palm, and Verchot (2005) who estimate the potential of agroforestry for carbon sequestration. In the case of forest conversion these changes cause an emission of 150 ± 75 Mg CO$_2$ ha^{-1}, while grassland rehabilitation leads to a fixation of 48 ± 24 Mg CO$_2$ ha^{-1} through the augmentation of soil carbon.

In organic soils the total carbon content depends primarily on the depth of the organic layer. In South East Asia peat profundity of several meters is common (Shimada, Takahashi, Haraguchi, & Kaneko, 2001), but the extent of a given thickness of peat in oil palm plantations is often inadequately known. These soils, usually waterlogged in their natural state, need to be drained prior to planting. Drainage of peat enhances decomposition of the organic matter by microbiological oxidation and leads to shrinkage through unalterable water loss. Both the decomposition and shrinkage processes are irreversible, lower the soil surface and can only be stopped by renewed flooding (Wösten, Ismail, & Wijk, 1997). In some areas agricultural land had to be abandoned as soil surface subsided to ground water level as a consequence of these processes (Mutert et al., 1999). Decomposition of peat leads to the emission of CO$_2$ and changes in the fluxes of N$_2$O and CH$_4$ balance (Inubushi, Furukawa, Hadi, Purnomo, & Tsuruta, 2003).

The IPCC gives annual rates for *carbon dioxide* emissions from decomposition of organic tropical soils. The default carbon loss under crop cultivation is quoted at 20 Mg C ha^{-1} a^{-1} (IPCC, 1997). This value is reconfirmed by measurements of the CO$_2$ flux on Malaysian peat that showed an annual carbon loss of up to 19 Mg C ha^{-1} (Murayama & Bakar, 1996b). It is assumed that the carbon loss through peat decomposition after drainage under pasture and forest is 25% of that under arable land (IPCC, 1997). Similar to pasture and forest in oil palm plantations, no ploughing or other peat decomposition enhancing soil preparation is needed. Hence, carbon release from peat under oil palm is in the magnitude of 5 Mg C ha^{-1} a^{-1}. This approximation does not, however, integrate the greater drainage depth needed for oil palm in comparison with the more shallow rooting arable crops. Therefore, it possibly underestimates the actual peat decomposition, because its rate is highest when the aerobic peat layer is thickest (Jauhiainen, Takahashi, Heikkinen, Martikainen, & Vasander, 2005).

In an overview of the world's major soils Driessen, Deckers, and Spaargaren (2001) give subsidence rates of as high as 10 cm per year for peat planted to horticultural crops, with liming and full fertiliser application; losses under a plantation

forest (*Acacia* sp.), with a closed canopy and much greater leaf fall than horticultural crops, are assumed negligible. Current research in oil palm plantations suggests, however, that most of the frond material decomposes on the soil surface and does not significantly affect the soil carbon pool (Fairhurst, 1996; Germer, 2003; Jourdan & Rey, 1997a). Wösten et al. (1997) found in long-term studies in an extensive cultivated peat area in Malaysia an annual surface subsidence rate of about 2 cm. The authors quantified that oxidation causes 60% and shrinkage 40% of surface subsidence. At a specific weight of peat of .1 g cm^{-3} and a carbon content of 60%, the authors determined an annual carbon loss by oxidation of 7.2 Mg ha^{-1}. In an overview of soil carbon density in forest soils of tropical Asia, Brown et al. (1993) used a bulk density of .15 g cm^{-3} and a carbon content of 45%. At the same rate of subsidence and oxidation as above, the values from this overview indicate an oxidative carbon loss of 8.1 Mg ha^{-1} a^{-1}. In an experiment in Johore, Peninsular Malaysia, the drainage of an agricultural field caused a net decomposition rate of 14.0 Mg ha^{-1} a^{-1} in neighbouring native forest (Kyuma, 2003). The magnitude of these C fluxes is confirmed through investigations by Melling, Hatano, and Goh (2005b), who measured a cumulative annual C emission from the peat soil under oil palm, caused by oxidation and root respiration, of 15 Mg ha^{-1}. From the reviewed references, a mean annual carbon loss by peat oxidation of 8.6 ± 3.8 Mg ha^{-1} is estimated, causing an emission of 31.4 ± 14.1 Mg CO_2 ha^{-1} per year.

Through the drainage of peat, the CH_4 emission from the soil surface is reduced and the soil turns gradually from a CH_4 source into a sink (Furukawa, Inubushi, Ali, Itang, & Tsuruta, 2005; Hadi et al., 2005), while the actual flux can be strongly influenced by climatic conditions (Inubushi et al., 2003). The IPCC guidelines suggest calculating the reduction of *methane* emissions caused by wetland drainage as follows: The area drained is multiplied by the difference in the average daily CH_4 emission rate before and after draining, which is multiplied by the number of days in a year the wetland was flooded. However, this work attempts to estimate the total fluxes from the oil palm ecosystem. Studying the CH_4 fluxes of different ecosystems on tropical peat, Melling, Hatano, and Goh (2005a) found a negative flux of – 3.6 ± 2.3 µg C m^{-1} h^{-1}. The annual cumulative oxidation was .2 kg of CH_4, which equals, at a global warming potential of 23, a net removal of 4.6 kg CO_{2eq} ha^{-1} per year through peat land drainage.

Quantitative information regarding *nitrous oxide* fluxes from natural and drained peat land in tropical regions is limited and very few measurements have been made in tree-based systems (Baggs, Cadisch, Verchot, Millarn, & Ndufa, 2002). Terry, Tate, and Duxbury (1981) measured in Florida an average N_2O emission of 99 g N_2O–N ha^{-1} d^{-1} from a drained Histosol planted to corn. Assuming a constant emission throughout the year, this is equal to 36 kg N_2O–N ha^{-1} a^{-1}. Hadi et al. (2000) found in Indonesia on peat under cassava (*Manihot esculenta* Crantz) cultivation a 20-fold higher N_2O emission than in secondary forest on the same soil type. On the cultivated fields they measured an emission of 144 g N_2O–N ha^{-1} d^{-1} over an average of 2 years. Under the assumption of a constant emission throughout the year this is equal to 53 kg N_2O–N ha^{-1} a^{-1}. In order to obtain the drainage-induced net emission of the land use change, emissions induced by nitrogen fertiliser must be deducted. The default factor in the IPCC guidelines for synthetic nitrogen fertiliser induced N_2O emission in the tropics is 10 kg N_2O–N ha^{-1} a^{-1} (IPCC, 1997). Peat decomposition under oil palm causes an annual N_2O flux between –1.0 and 3.1 kg N_2O–N ha^{-1} when the fertiliser-induced emission is omitted and, based on the

decomposition rate, the N_2O emission is assumed to be 25% of that under upland field crops. This is well below the default emission value of 10 kg N_2O–N ha^{-1} per year as suggested by the IPCC for Histosol in tropical regions (IPCC, 1997). Integrating all reviewed data, the net emission is 4.1 ± 5.5 kg N_2O–N ha^{-1}. At a climate activity of N_2O that is 296 times higher than that of CO_2, the annual N_2O emission from drained peat equals 1.2 ± 1.6 Mg CO_{2eq} ha^{-1}.

To sum up, the total potential annual emission of CO_2 and N_2O from decomposition and sequestration of CH_4 through drainage of tropical peat soil is 33 ± 16 Mg CO_{2eq} per hectare. This corresponds to a total emission of 816 ± 393 Mg CO_{2eq} per hectare within a 25-year economic life span of oil palm.

7 Carbon fixation in oil palm plantation biomass

The amount of carbon bound in oil palm plantation biomass is primarily a function of palm growth and the understorey. Published values on oil palm AGB range from 50 Mg ha^{-1} to over 100 Mg ha^{-1} towards the end of the plantation's economical live span after 25 years. Most of these figures represent the biomass at a certain palm age without stating actual and maximum trunk height. As timing for replanting is rather determined by palm height than by age, the suitability of these data to model biomass accumulation is limited. Further, the reviewed references often state the biomass without information on planting density and local environment, to both of which the vegetative development of oil palms responds distinctly (Germer & Sauerborn, 2004; Henson, 1998; Henson & Dolmat, 2003). The available data also contain inherent uncertainties associated with the lack of standardised methods. While some researchers measure the total biomass including the understorey, others consider only the palms. Some of the latter exclude the productive organs. However, the single most important shortcoming lies in the fact that it is not always clear whether the biomass includes the leaf bases attached to the trunk (Corley & Tinker, 2003).

The typical biomass accumulation in tree plantations follows a curve of quick initial growth and thereafter a minor increase (Sanchez, 2000). A linear equation to calculate the carbon stocks as provided in ''Good Practice Guidance for Land Use, Land-use Change and Forestry'' by the IPCC (2003) tends to underestimate actual values. Comparing a logarithmic growth function of palms aged 3–25 years obtained through destructive sampling in Sumatra (Syahrinudin, 2005) with the linear approach, the latter underestimates the time-averaged biomass by 23%. Biomass data from 51 oil palm fields (Corley & Tinker, 2003; Haron, Zakariae, & Anderson, 1999b, c; Henson, 1998; Henson & Chai, 1997; Henson & Dolmat, 2003; Husin, Hassan, & Mohammed, 1986; Lasco, 2002; Palm et al., 2000; Ross, 1999; Syahrinudin, 2005; Thenkabail et al., 2004) were plotted in Fig. 1. Integration of the fitted equation returns a time-averaged oil palm AGB of 60 ± 20 Mg ha^{-1} for 25 years after planting, where the standard deviation is an estimation taking the large variation of the plotted data into account.

The root biomass of oil palm increases with the AGB, while its maximum volume depends strongly on soil properties and water availability. In West Sumatra, Fairhurst (1996) found 7 Mg ha^{-1} of dry root biomass in the upper .4 m soil layer under 13-year-old palms. Under 23-year-old palms in West Malaysia, Khalid, Zin, and Anderson (1999) calculated a root biomass of 16 Mg ha^{-1} to a soil depth of .6 m.

Fig. 1 Above ground oil palm
biomass – potential vs. linear
estimation of time-averaged
biomass. The thin solid
horizontal line indicates the
time-averaged AGB as
obtained by integration of the
potential curve; the dashed
horizontal line represents the
time-averaged biomass of the
linear curve respectively
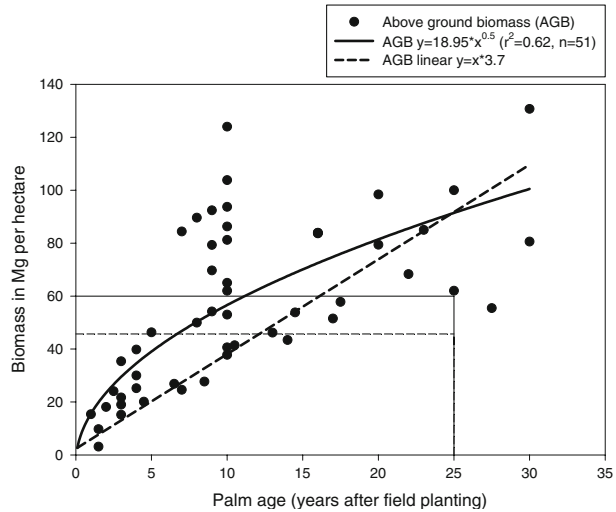
Most of the oil palm root biomass is usually present in the uppermost .4 m of the soil profile (Jourdan & Rey, 1997b), while under less favourable environments a significant part of the roots may penetrate to greater depth. In West Africa, Braconnier and Caliman (1989) measured a root biomass of 59 Mg ha^{-1} on degraded highly leached soil to a depth of 6 m under oil palms aged 16 years. The discrepancy of depth considered and methods applied in oil palm BGB measurements diminishes the aptness to develop age or AGB related growth curves. Linear equations developed by Henson and Dolmat (2003) and Syahrinudin (2005) for the assessment of BGB based on AGB show a high coefficient of determination for adult palms, but overestimate the root biomass in young plantings. The time-averaged oil palm root biomass of 20 ± 5 Mg ha^{-1} used for the greenhouse gas balances is an approximation based on all reviewed references.

Biomass of ground cover vegetation decreases with palm growth and heavier shade. In fully established *Pueraria phaseoloides* Benth. dominated cover, Ross (1999) measured an AGB of 8.6–9.3 Mg ha^{-1}, including 27% dry leaves. The necromass can account for over half of the AGB. Khalid, Zakaria, and Anderson (2000) found 56% dry material in a total of 11.4 Mg ha^{-1} ground cover AGB under 18-month-old palms. Under light shade plantation, understorey accumulates an AGB of about 4 Mg ha^{-1} (Ezenwa, Aribisala, & Aken'ova, 1996), whereas under a closed canopy the AGB may be less than 1 Mg ha^{-1} (Parrotta, 1992). The root biomass of the undergrowth is estimated by applying a shoot to root ratio of .3 for non-forest vegetation (Fearnside, 2000). Assuming a fast ground cover establishment with a maximum AGB of 10 Mg ha^{-1} and a simplified linear biomass loss through increased shading to 1 Mg ha^{-1} at canopy closure at 5 years after planting (Germer & Sauerborn, 2004; Gerritsma & Soebagyo, 1999), the time-averaged total ground cover biomass is 2.5 ± 1.0 Mg ha^{-1}.

The above calculation of the time-averaged oil palm AGB assumes a complete and quick decomposition of oil palm residues (Haron et al., 1999a; Khalid, Zin, & Anderson, 2000) at completion of the 25-year economic life span. If, contrariwise, these residues are burnt at an efficiency of 40% (Ross, 1999), the application of the emission ratios for open burning of cleared forest (IPCC, 1997) results in a reduction

of the time-averaged carbon stock in the palm biomass by a negligible .2%. The total time-averaged above and BGB in an oil palm plantation is 82.5 ± 26.0 Mg ha^{-1}. Considering a carbon content of 40.4% for oil palm biomass (Syahrinudin, 2005) and of 50% for the remaining vegetation (IPCC, 1997), both palms and understorey together fix 35.3 ± 11.0 Mg of carbon per hectare within the economic life span of oil palm. Accordingly, a time-averaged total of 129.3 ± 40.3 Mg CO_2 per hectare is assimilated.

8 Greenhouse gas balance

Establishment of oil palm plantations causes emission of GHGs by biomass burning and decomposition, loss of soil carbon and decomposition of peat. On the other hand, plantation biomass and accumulation of soil organic matter act as a sink for atmospheric carbon. The GHG fluxes from the principal sources and sinks initiated by conversion of grassland and forest into oil palm plantations are balanced in Table 3. The conversion of forest into oil palm plantations causes an emission in a range of less than 650 to over 1,300 Mg ha^{-1}, depending on soil type, within the considered 25-year time frame. Despite the large potential deviation, the emission values calculated emphasise that conversion of forest is in any case a significant source of GHG. Decomposition of organic matter in peat soil is likely to exceed GHG emissions derived from forest biomass. In contrast, establishment of oil palm plantations on grassland result in an increase in biomass and soil carbon. These carbon sinks not only neutralise emissions caused by land conversion, but lead to a net sequestration of 135 Mg ha^{-1} CO_{2eq}. In all three scenarios considered, renunciation of biomass burning in the process of land clearing offers only a minute potential to reduce GHG emissions. In the case of grassland rehabilitation burning increases carbon dioxide equivalent emission by less than 1% and in the case of forest conversion up to 3%. Nonetheless, the detriment of burning is obvious: wild life extinction, harmful smoke, erosion and nutrient losses (Levine, Bobbe, Ray, Singh, & Witt, 1999; Rowell & Moore, 2000).

Table 3 Green house gas balance in carbon dioxide equivalents (Mg ha^{-1}) for oil palm plantation establishment on degraded grassland and on forest on mineral and organic soil

	Land clearing	Change in soil carbon or peat decomposition	Fixation in oil palm plantation biomass	Balance
Grassland rehabilitation (zero burning)	42 ± 27	-48 ± 24	-129 ± 40	-136 ± 37
Grassland rehabilitation (burning)	43 ± 28	-48 ± 24	-129 ± 40	-134 ± 36
Forest conversion (zero burning) on mineral soil	627 ± 326	150 ± 75	-129 ± 40	647 ± 361
Forest conversion (burning) on mineral soil	648 ± 337	150 ± 75	-129 ± 40	668 ± 372
Forest conversion (zero burning) on peat	627 ± 326	816 ± 393	-129 ± 40	1314 ± 679
Forest conversion (burning) on peat	648 ± 337	816 ± 393	-129 ± 40	1335 ± 690

Minor GHG fluxes affected by land conversion, indirectly influenced sources and sinks as well as changes in the lower soil profile were not taken into account due to uncertainty in the dynamics and volumes involved. The integration of the following points requires additional information, but may eventually achieve further refinement of the emission balance:

- The carbon stocks in deeper soil layers are also apt to undergo reduction or incrementation in response to deforestation and plantation establishment. Though probably a rather slow process, this might ultimately contribute significantly to the GHG balance (Fearnside & Barbosa, 1998).
- To a certain extent, most forests suitable for oil palm plantation establishment have already suffered biomass loss through human activity. The actual C stocks of these forests are below the potential as defined by the environment (Brown et al., 1993). For this reason, and because of their natural C sequestration potential and that enhanced through CO_2 fertilisation, the forests themselves may act as a significant C sink (Phillips et al., 1998).
- Biomass burning is also a source of carbon monoxide, non-methane volatile organic compounds, nitrogen oxides and ozone (Andreae, 1991). The IPCC provides guidelines for the calculation of nitrogen oxides emission from biomass burning. There is, however, no agreement on the global warming potential of any of these criteria pollutants, nor were they included in 'Annex A' of the Kyoto Protocol.
- In the assessment a constant rate of peat subsidence is assumed, although initial subsidence after drainage is much higher (Mutert et al., 1999; Wösten et al., 1997). Furthermore, if fire is used in land clearing, a notable part of the peat burns or smoulders as well (Muraleedharan, Radojevic, Waugh, & Caruana, 2000) and the ash produced enhances peat decomposition (Murayama & Bakar, 1996a).
- The indirect impact of land use change on the environment is a further potential source of GHGs. Most obvious is the impact on the immediate surroundings of a new planting. Fragmentation and destruction of forest, e.g., due to road and drain building, as well as urbanisation result in biomass loss and GHG emission. An indirect impact through air and water pollution is likely to occur in a much greater radius. The atmospheric deposition of air pollutants can damage forests and thus reduce the amount of carbon stored (IPCC, 1997). Thick haze caused by biomass burning as during the 1997/1998 El Niño event in Southeast Asia possibly reduces photosynthesis and carbon fixation (Davies & Unam, 1999). Further, land clearing increases runoff, erosion and the sediment loading of rivers. Deposit of these sediments on reefs results in smothering of the corals through reduced light levels and over-nutrition (Bryant, Burke, Mc Manus, & Spalding, 1998; Wesseling, Uychiaoco, Alino, Aurin, & Vermaat, 1999) and might affect the coral reefs as a potential long-term carbon sink.

Though important, the integration of these points in the emission calculations is not expected to alter the magnitude of the results significantly. Extending the time frame considered, however, distinctly increases the emission from plantation establishment through peat forest conversion. In the current assessment decomposition of peat is regarded throughout the economic life span of oil palm of 25 years, which limits soil surface subsidence. Continuing cultivation would lead to an

additional surface subsidence of .5 m and an emission of 818 Mg CO_{2eq} per hectare every 25 years until peat decomposition is complete. Mineral soil, on the other hand, is assumed to reach new soil carbon equilibrium within 20 years following alteration of land use (IPCC, 1997). Thus, the conversion of forests on organic soil initiates a GHG flux that might continue for several oil palm rotations, whereas on mineral soil no further net change in soil carbon is to be expected after the first rotation.

9 Conclusion

The calculations of CO_{2eq} emissions caused and carbon sequestered in response to oil palm establishment clearly show the advantage of grassland rehabilitation over forest clearance. Regulations and incentives steering the expansion of the oil palm industry are thus potential tools to reduce emissions through prevention of new plantation establishment in forest areas, especially on organic soils. In addition, promoting grassland rehabilitation for oil palm plantings could contribute to carbon sequestration. The 'United Nations Framework Convention on Climate Change' flexibility mechanism could offset additional costs for plantation establishment on grassland. The purpose of these mechanisms is to assist developed countries in pursuing least-cost options to meet their target commitments through the generation of certified credits from projects undertaken in developing countries. While details of implementation are yet to be settled, it may involve the establishment of plantations. With global emission trading becoming a reality, emitters unable to meet their own targets could pay off through carbon sequestration in oil palm plantations. Currently the price for carbon dioxide emission credits traded on the European market is above 27 Euro per metric tonne (PointCarbon, 2006). At this price the rehabilitation of grassland through establishment of oil palm plantations would value above 4.000 Euro per hectare.

Besides the refinement of the primary data used in the emission estimates, there is a need for a detailed cost benefit analysis of oil palm plantation establishment on rehabilitated grassland. Also, the secondary impacts on local communities, biodiversity and environment need further assessment.

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