Carbon dioxide emission assessment associated with land use transitions to agriculture is important for designing strategies in reducing greenhouse gas emission. Net CO₂ emissions is the sum of 1) the above ground biomass C loss of the initial land use because of land clearing, 2) the above ground C accumulation by the subsequent plantation crops, 3) soil organic matter decomposition, and 4) soil organic C burning if land management or land clearing involves fire. For mineral soils, in most cases, conversion of primary and secondary forests with time average C stocks of about 200 and 132 t/ha, respectively, results in a net C emission. However, if shrub or Imperata grassland, with respective C stocks of 15 and 2 t/ha is rehabilitated to plantation, it generally results in a net C sequestration.

For peat soil, CO₂ emission is caused by peat decomposition, peat burning (if any), and the aboveground C decomposition and/or burning. Rehabilitating peat shrub (with assumed C stock of about 15 t/ha and average drainage depth of 40 cm) instead of clearing peat forest (with assumed C stock of about 200 t/ha and drainage depth of 0) for agriculture reduces CO₂ emissions of about 862 t CO₂-e/ha/25 year (34 t CO₂-e/ha/year) because of substantial reduction in the plant biomass and possible peat soil carbon loss due to burning. Peat shrub remains as peat shrub emits about 22 t CO₂-e/ha/year. If peat shrub is rehabilitated to paddy field, rubber plantation or oil palm plantation, the emission levels become 11, 7, and 30 t CO₂-e/ha/year, respectively. This means that rehabilitating peat shrub to paddy field or rubber plantation, results in net emission reduction of 11 and 15 t CO₂-e/ha/year, respectively, whereas rehabilitating it to oil palm plantation increases net emission of only 8 t CO₂-e/ha/year, relative to leaving the peat shrub as is. Therefore, new plantation development should be prioritized on mineral soils’ shrub and Imperata grasslands or on peat shrub as these conversions, in most cases, result in the net CO₂ sequestration and potentially improve the livelihood of the communities.

Keywords: Carbon dioxide emission, carbon sequestration, land use transition, plantations, peatlands, mineral soils
Plantation or estate crops in Indonesia contribute to as high as 15% of US$ 541,593 billion of agriculture and forestry gross domestic products (GDP) or 2% of US$ 3,949,321 billion total national GDP in 2007 (BPS 2009). More than 50% of plantation GDP was generated from oil palm plantation.

The demand for plantation products, especially for bio-fuels has been and will continually be increasing with the scarcity and, at the same time, skyrocketing prices of fossil fuels. Areas of oil palm plantation, for example, has increased dramatically at a rate of 12.3% annually from only 290,000 ha in 1980 to 6,075,000 ha in 2006 (Ditjenbun 2007; IPOB 2007). Currently, a total of about 17 million ha land has been utilized for rubber, coconut, oil palm, coffee, cacao, tea, and pepper plantations (BPS 2006).

There are several types of potential land that can be converted to plantations. These include primary forest, secondary forest, shrub, and Imperata grassland. The recent estimate is that, in total, about 15.3 million ha more lands throughout Indonesia are potentially available for the development of plantations (Mulyani and Las 2008).

However, despite the economic and livelihood importance, extensification of plantation is feared to cause environmental degradation. Plantation extensification has been perceived as the main cause of deforestation and that deforestation and land conversion to plantation has been perceived as the main source of Land Use, Land-Use Change and Forestry (LULUCF) related CO₂ emissions (e.g. Hooijer et al. 2006; WWF Indonesia 2008), although in reality, not all of deforestation are intended for plantation or agriculture development. In Riau, for example, only about 50% of the deforestation ended up in plantation (mainly acacia and oil palm plantations, with acacia plantation took the larger share) (WWF Indonesia 2008), while the rests change into secondary forest or shrubland. Most of this ‘degraded’ or wasted shrubland could potentially be used for some kinds of plantation provided that the supporting infrastructures, institutions, and markets are conducive for such.

Extensification of plantation occurs both on mineral and peat lands. Recently, the extensification on peatland accelerates (WWF Indonesia 2008) because of relatively lesser tenure conflicts on these lands. There are similarities and differences in the process by which carbon is emitted or sequestered between the two soil types, while the processes of above-ground CO₂ emissions are common for both mineral and peat soils. The process of decomposition of peat is very different from that of mineral soil. The rate of CO₂ emissions from peat decomposition can be orders of magnitude higher than that of mineral soils (Agus et al. 2007). However, despite the lower emission rates per unit area, the large proportion of land makes this assessment important both on mineral and peatlands.

Hooijer et al. (2006) estimated that the tropical peatlands, 80% of which are distributed in Indonesia, contribute to as high as 2 Gt (giga tonnes) CO₂ emissions annually; 1.4 Gt of which is associated with peat forest fire (wild fire and deforestation related) and 0.6 Gt is associated with peat decomposition. This high carbon loss can be reduced if the country prioritizes the extensification on idle lands with low carbon stock such as grassland or shrubland. Analysis by Belllassen and Gitz (2008) for example, evaluated the possibility of avoided deforestation in Cameroon and found that carbon credit of US$2.85 for abatement of each ton of CO₂ would save the current natural forest from conversion, while at the same time reduces emissions from deforestation.

Carbon budget analysis deems essential as an input for policy makers in regulating mitigation strategies associated with plantation extensification. This analysis may also form a basis for calculation in the carbon trading post Kyoto Protocol starting in 2012, through the scheme called “Reducing Emissions from Deforestation and Degradation” (REDD). This paper focuses on the estimate of carbon budget as forest, shrubland or Imperata grassland is converted into plantation. The time frame for calculation starts from the initial land use system (forest, shrub or grass-land) until the completion of one economic cycle of most plantations of 25 years.

CARBON LOSSES AND GAINS ASSOCIATED WITH LAND USE CHANGE

When an ecosystem is transformed to plantation, green house gasses, especially CO₂ emissions occur during land clearing and land preparation through biomass burning and/or decomposition. The amount of C stock of the biomass of initial land use determines the amount of CO₂ emissions associated with land clearing and land preparation.

During the clearing of mineral soil, C loss mainly associates with plant biomass and necromass burning and decompositions. On peatland, burning (if applicable) affects the aboveground biomass as well as the abundant organic matter of peat soil. Recently, however, the government of Indonesia has banned burning and this it is expected that burning effects will substantially decrease in the future.

After the plantation crops are planted, the main process of aboveground carbon stock change is C sequestration by the crops. On mineral soils, little change occurs in soil C stock during the crop period, but on peat soils substantial amount of C is lost from peat decomposition mainly due to drainage.

Carbon Pools on Mineral Land

Carbon on mineral land is stored in plant biomass (shoot and root), necromass, and below ground (soil carbon). The total amount and proportion of C in each of these three storages varies depending on soil fertility, climate, and land use types.

Necromass is all kinds of dead, non-decomposed plant parts including tree stumps, standing dead stems, branches, twigs and leaves. Its C storage is relatively high (ranging from 30 to 80 t C/ha) in tree based systems, but negligible in annual cropland and Imperata grassland (Tomich et al. 1998) as well as on intensive tree based systems such as tea plantation. In this analysis, we ignore the contribution of necromass because of high uncertainty and insufficient supporting literature for the systems under the current discussion. Thus only plant biomass and soil carbon are taken into account for land use transitions on mineral lands.

Carbon in plant biomass

The amount of carbon stored in the plant tissues of each land use varies (Table 1) depending on soil fertility, climatic conditions, elevation, and the plant growth stage. Carbon stocks are expressed in terms of time average, i.e., based on the average of time course over the rotation
of the commodity. Figure 1 demonstrates graphically the time course carbon stock and time average carbon stock. This method of time average representation allows the comparison of carbon stocks in systems that have different tree-growth and harvesting rotations. This method is similar to the average storage method described in the IPCC Special Report on LULUCF (Watson et al. 2000).

Table 1 provides the bases of calculation of above ground change in carbon stock between the initial land use system and the subsequent plantation system.

Table 1. Time-averaged C stock of selected land use systems used in the current estimate.

<table>
<thead>
<tr>
<th>Land use systems</th>
<th>Time average C (t/ha)</th>
<th>Reference, remarks</th>
</tr>
</thead>
<tbody>
<tr>
<td>Primary forest (Indonesia)</td>
<td>300</td>
<td>Palm et al. (1999)</td>
</tr>
<tr>
<td>Secondary forest (Central Kalimantan, Indonesia)</td>
<td>132</td>
<td>Brearly et al. (2004)</td>
</tr>
<tr>
<td>Shrub</td>
<td>15</td>
<td>Prasetyo et al. (2000)</td>
</tr>
<tr>
<td>Imperata grassland</td>
<td>2</td>
<td>Palm et al. (2004)</td>
</tr>
<tr>
<td>Oil palm (Indonesia)</td>
<td>60</td>
<td>Recalculated from Rogi (2002)</td>
</tr>
<tr>
<td>Rubber agroforest (Indonesia)</td>
<td>68</td>
<td>Averaged from Palm et al. (2004)</td>
</tr>
<tr>
<td>Coconut</td>
<td>60</td>
<td>Adjusted from 98 t/ha according to IPCC (2006)</td>
</tr>
<tr>
<td>Jatropha</td>
<td>10</td>
<td>June et al. (2008) based on Niklas (1994)</td>
</tr>
<tr>
<td>Tea</td>
<td>28</td>
<td>Adapted from Kamau et al. (2008)</td>
</tr>
<tr>
<td>Sugar cane</td>
<td>9</td>
<td>Soejono (2004), modified for shaded coffee</td>
</tr>
<tr>
<td>Coffee</td>
<td>51</td>
<td>Hairiah and Rahayu (2007) for shaded coffee</td>
</tr>
<tr>
<td>Cacao</td>
<td>58</td>
<td>Lasco (2002); IPCC (2006)</td>
</tr>
</tbody>
</table>

In all cases, the error is around 20–40%.

1Recent value from ICRAF (unpublished) is around 40 t C/ha.

Figure 1. Carbon content in the oil palm plant biomass with respect to age and the time average C (adapted from Rogi 2002).

Time average carbon stocks of initial land uses as high as 300, 132, 15, and 2 t/ha, respectively for forest, secondary forest, shrub, and Imperata grassland, were chosen for the current calculation.

Root biomass is more difficult to estimate and its measurement requires destructive sampling. For tropical rain forest, Hairiah et al. (2001) suggested the proportion of above and below ground biomass of 4:1 for upland system, 10:1 for wetland, and 1:1 for marginal land. For simplification of the calculation, we use the above-below ground proportions of 4:1 for upland and 8:1 for drained peatland. In most of published data, the estimate of root C is included in the presentation of biomass C, unless otherwise stated.

Time average C for primary forest in the tropics ranges from 207 to 405 t/ha. This varies depending on the soil and climatic conditions (Palm et al. 1999). We choose the mid value of 300 t/ha in this analysis. Secondary forests in Kalimantan have C stock of around 132 t/ha (Brearly et al. 2004) and this value is adopted in our analysis.

Kamau et al. (2008) measured carbon stock based on destructive sampling of 14- and 29-year clonal, and 43- and 76-year old seedling tea plantations. They found that total C stocks amounted to 44 and 72, and 43 and 69 t C/ha for clonal and seedling bushes, respectively. Despite the relatively young clonal plantation, the higher crop density with plant spacings of 1.22 m x 0.61 m in the clonal compared to 1.22 m x 1.22 m under the seedling plantation, lead to somewhat higher C stock in the clonal plantation. We use the time average C stock value of 28 t/ha, based on half of the mid values of C stock in the clonal and the seedling tea plantations.

Soil carbon

Carbon in soil exists in organic and inorganic forms. Most of inorganic forms are in calcium and/or magnesium carbonates. In calcareous soils, the amount of inorganic C may exceed that of the organic one. However, not all soils contain inorganic C because of dissolution of carbonates during soil formation (Nelson and Sommers 1982). Because of the relatively small amount of inorganic soil C, except for calcareous soils, the discussion of soil C in this paper concentrates on organic C.

Carbon stock in the 0–30 cm depth of the humid tropical forest ranges from 5 to 180 t/ha (IPCC 1997). Based on this range, Germer and Sauerborn (2008) suggested the carbon stock for the humid tropical soils suitable for oil palm plantation as high as 120 ± 60 t/ha.

The change in soil carbon content is determined by factors such as soil tillage and organic matter inputs. Murty et al. (2002) suggested a value of 30% for soil organic matter reduction as forest is
converted to plantation. Therefore, with the assumed initial carbon stock of 120 ± 60 t/ha in the forest soil, the reduction will be about 40.8 ± 20.4 t/ha when the land is converted to plantation. On the other hand, when Imperata grassland is converted to plantation, then an increase in soil carbon stock of as high as 13.2 ± 6.6 t/ha from the initial stock of 40.8 ± 20.4 t/ha is expected.

**Change in C Stock on Peatland**

**Above ground biomass**

The processes of above ground (biomass and necromass) change in carbon stock in peatland is similar to that of mineral land, but the values of biomass C stock in peatland, in general, is somewhat lower than that of mineral soil. Table 2 provides the above ground C stock values used in this current estimate.

**Change in C stock of peat soil**

The largest stock of carbon in peat land is in the belowground peat itself. One meter layer of peat can store more than double C stock of plant biomass of peat forest (300–700 t/ha in the peat versus 100–200 t/ha in the plant biomass) (Agus and Subiksa 2008). If land clearing involves fire, this high carbon stock in the peat is subject to burning along with the burning of plant biomass and thus contribute to higher GHG emissions. Land use change from peat forest to plantation, especially for those plantations requiring a relatively deep drainage will change the function of the former as a carbon sequester to carbon emitter (Hooijer et al. 2006; Agus and Subiksa 2008).

Drainage depth is the main factor determining the rate of carbon emissions. With a relatively deep drainage (about 60 cm) of oil palm plantation, using the drainage depth-emission rate relationship as suggested by Hooijer et al. (2006), the estimated emission is about 54 t CO₂/ha/year. This value is much lower than IPCC (1997) suggested value of 20 t C/ha/year, or 73.4 t CO₂-e/ha/year for peatland planted to agricultural crops. Melling et al. (2005) based on measurement in North Kalimantan and Kyuma (2003), from the measurement in Johor, Malaysia Peninsula, found the values of 15 and 14 t C/ha/year or 53 and 51 t CO₂/ha/year, respectively, which were comparable to that of Hooijer et al. (2006). The lower estimate of 26.4 t CO₂/ha/year was proposed by Wöstén et al. (1997) for peat in Sarawak, Malaysia, used for various kinds of agricultural systems, but no specification of drainage depth was described.

**CALCULATION OF CARBON BUDGET**

The estimate of the net carbon emissions is based on the relationship:

\[ E = E_a + E_{bd} + E_{bo} - S_a \]  \[1\]

where \( E_a \) is emission from the above-ground biomass burning, \( E_{bd} \) is the emissions from belowground (peat soil) burning during deforestation, \( E_{bo} \) is emission from belowground peat oxidation, and \( S_a \) is sequestration of CO₂ from the atmosphere into plantation crop biomass through photosynthesis.

When an ecosystem of forest, secondary forest, shrub or Imperata grassland is converted to plantation, to comply with IPCC Guidelines, we assume that 100% of the plant biomass is burned or decomposed rapidly and thus contribute to atmospheric CO₂.

\[ E_a = C_b \times 3.67 \]  \[2\]

where \( C_b \) is carbon stock in the biomass of initial land use in t/ha. The coefficient 3.67 is the conversion factor from C to CO₂ based on atomic weights of C and O of 12 and 16 g, respectively.

The volume of peat burned is determined by the peat water content and water table depth, as well as the amount of fuel from the plant biomass that trigger the burning. Peat soil undergoes burning of as deep as about 50 cm during El Niño years (Page et al. 2002) or 30 cm during normal years (R. Hatano, pers. comm.). Our observation during the normal year in several places showed a wide range of burning depth from zero to around 30 cm. In this analysis, we assume that when peat forest is cleared, 15 cm of peat layer is burned and we believe that this is a liberal assumption. If the initial ecosystem is peat shrub, we assume that only about 5 cm of peat is burned during land clearing. For mineral soil, the amount of soil C burned is negligible.

\[ E_{bd} = V_p \times C_d \times 3.67 \]  \[3\]

where \( V_p \) is the volume of peat burned (m³) and \( C_d \) is the volume based peat C content or the mass of C per unit volume of soil (t/m³).

\[ C_d = BD \times C_r \]  \[4\]

where \( BD \) is peat or soil bulk density which is the oven dry weight of the soil per its unit volume (t/m³) and \( C_r \) is the fraction of carbon in the soil mass (t/t or frequently expressed as percentage).

Literatures vary in the estimate of emissions from peat decomposition, \( E_{bo} \), depending on measurement method and environmental conditions. In a review of several research results, Hooijer et al. (2006) developed a linear relationship that, for the drainage depth ranging from 30 to 120 cm, emissions due to peat decomposition increases as much as 0.91 t CO₂/ha/year for every centimeter increase in drainage depth. However, the data reviewed by

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**Table 2. Assumptions used in carbon budget calculation for peatland forest, shrub and plantation/farming systems.**

<table>
<thead>
<tr>
<th>Aspects</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Time average C stock in the biomass of peat forest (t/ha)</td>
<td>200</td>
</tr>
<tr>
<td>Time average C stock in the biomass of peat shrub (t/ha)</td>
<td>15</td>
</tr>
<tr>
<td>Time average C stock in the biomass of rice (t/ha)</td>
<td>2</td>
</tr>
<tr>
<td>Peat carbon density (t/m³)</td>
<td>0.05</td>
</tr>
<tr>
<td>Thickness of peat layer burned during peat forest deforestation (cm)</td>
<td>15</td>
</tr>
<tr>
<td>Thickness of peat burned during peat shrub clearing (cm)</td>
<td>5</td>
</tr>
<tr>
<td>Average drainage depth of peat forest (cm)</td>
<td>0</td>
</tr>
<tr>
<td>Average drainage depth of peat shrub (cm)</td>
<td>40</td>
</tr>
<tr>
<td>Average drainage depth of oil palm plantation (cm)</td>
<td>60</td>
</tr>
<tr>
<td>Average drainage depth of rubber plantation (cm)</td>
<td>20</td>
</tr>
<tr>
<td>Average drainage depth of paddy field (cm)</td>
<td>10</td>
</tr>
</tbody>
</table>

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Hooijer et al. (2006) were based on several experiments using closed chamber from which there were no separation between autotrophic root respiration and microbial decomposition. Knorr et al. (2008), based on research in temperate zone, found that 55–65% of peat respiration was generated by root respiration. Since CO₂ generated by root is offset by atmospheric CO₂ through photosynthesis, thus only about 35–45% of the soil respiration associated with peat decomposition contributing to the green house gas. Mäkiranta et al. (2008) found the contribution of heterotrophic decomposition of as high as 42%, root respiration as high as 41%, and above-ground litter decomposition as high as 17%. Our study in Aceh, Sumatra, found that CO₂ emitted from the non-rooted zone was about 62% of that of the rooted zone (Handayani 2009), indicating a significant contribution of the root to the production of CO₂. Therefore, in this calculation, for the estimation of emission from peat decomposition, the relationship of Hooijer et al. (2006) is corrected by a factor of 0.6.

Sequestration in plant biomass,

\[ S_x = C_p \times 3.67 \]  \hspace{1cm} [5]

where \( C_p \) is time average carbon stock in the plant biomass of the plantation system with the values as listed in Tables 1 and 2 for mineral and peatlands, respectively.

ESTIMATES OF CO₂ EMISSIONS

Carbon Budget on Mineral Land

Using the \( E_a \) and \( S_0 \) terms of Equation [1], the estimated annual average net CO₂ emission associated with various land use changes into plantation is presented in Figure 2. Conversion of primary forest to plantation results in the average CO₂ emissions ranging from 40 t/ha/year for rubber to 49 t/ha/year for sugar cane because sequestration by the plantation crop biomass is too small to compensate the loss carbon from the initial land use biomass burning and/or decomposition. This calculation included emission from the soil of about 5.9 t CO₂-e/ha/year. Conversion of secondary forest to oil palm, coconut, rubber, coffee agro-forestry, or cacao results in the net CO₂ emission of less than 12 t/ha/year. Conversion of secondary forest to Jatropha, tea or sugar cane, results in a much higher CO₂ emission ranging from 15 to 18 t/ha/year due to the low time average carbon stock of these plantation crops.

Shrub and grasslands with the estimated total area of over 11 million ha (Garrity et al. 1996) are potentially be converted to plantation in such a way that the pressure to convert forests will be reduced and the chance to sequestrate carbon will increase. Using these two types of initial ecosystem for plantation extensification results in net negative emissions. Infrastructural, socio economic and tenure constraints for such carbon efficient conversion should be overcome by the responsible institutions for this C sequestration to take place.

\[ \text{Sequestration in plant biomass,} \]

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Carbon Budget from Peatland

Using Equation [1] and assumptions as listed in Table 2, carbon budget resulted from forest transition to oil palm plantation, rubber plantation or paddy farming in peatland is presented in Figure 3. With the influence of drainage depth, oil palm plantation with the deepest drainage depth emits the highest amount (1,600 t/ha per 25-year cycle of oil palm or the average of about 64 t CO2/ha/year) of net CO2 relative to that of other land use systems. The highest component of emission under oil palm plantation is from peat decomposition which amounts 33 t/ha/year (Figure 3). The second most important component is emission from burning and/or decomposition of plant biomass. The complete burning of peat forest biomass with C stock of 200 t/ha (Rahayu et al. 2005) results in 734 t CO2-e/ha emission. This process likely occurs only once, during land preparation. If the value of emission from burning is averaged over the plantation period of 25 years, it results in 29.4 t CO2-e/ha/year. The third component is the burning of peat layer which is assumed as thick as 15 cm for peat forest as the initial land use. With the assumed volume based carbon content of 0.05 t/m3, using Equation [3], then the estimated amount of emissions from burned peat becomes 3 t C/ha/year or 11 t CO2-e/ha/year.

Carbon sequestration by the plantation crops occurs during the plant tissue formation through the plant photosynthesis. Based on Figure 3, the amount of sequestration by oil palm, calculated using Equation [5] is as high as 2.4 t C/ha/year or equivalent with 8.8 t CO2/ha/year. Taking into account all components of emissions and sequestration, on the basis of one cycle of oil palm growth, the estimated net emission rate for forest transition to oil palm plantation is about 64 t CO2-e/ha/year. The shallow depths of drainage for rubber and paddy resulted in net annual C loss of 11.3 and 12.4 t/ha/year or an equivalent of 41 and 45 t CO2/ha/year, respectively.

If peat shrub, instead of peat forest is used for plantation extensification, the amount of emissions from biomass decomposition and/or burning, peat burning, as well as peat decomposition will be much smaller, while sequestration through plant biomass growth likely remains the same as that with peat forest as the initial land use. Peat shrub, with assumed drainage depth of about 40 cm, emits about 22 t CO2-e/ha/year from peat decomposition. If the peat shrub is rehabilitated to paddy field or rubber plantation, the drainage depths become 10 and 20 cm and emissions from decomposition likely decrease to 6 and 11 t CO2-e/ha/year, while the net CO2 emissions become 11 and 7 t CO2-e/ha/year respectively (Figure 4). This means that this type of rehabilitation results in a net emission reduction of 11 and 15 t CO2-e/ha/year, respectively. In the case of peat shrub rehabilitation to oil palm plantation that requires drainage depth of about 60 cm, the peat decomposition component likely increases to about 33 t CO2-e/ha/year while the net emission becomes 30 t CO2-e/ha/year. This means that rehabilitation of peat shrub to oil palm plantation only increases emission as much as 8 t CO2-e/ha/year, relative to leaving the peat shrub as is.

CONCLUSIONS AND IMPLICATIONS

On mineral soils, the net greenhouse gas emissions from the transition of a certain land use/land cover system to plantation is largely determined by the difference in the aboveground carbon stock between the initial land use systems and the subsequent plantation crops. Greenhouse gas emissions could be reduced either through avoided deforestation and/or establishment of high carbon stock plantation systems on low carbon stock ecosystems such as idle Imperata grasslands or shrub land.

For peatland, the amount of CO2 emission from peat decomposition ranges from

Figure 3. Estimated CO2 emission from peat forest transition to agricultural systems.

Figure 4. Estimated CO2 emissions from the transition of peat shrub to agricultural systems (dash line is peat decomposition level if shrubland remains shrubland).
zero in primary peat forest to more than 30 t CO₂/ha/year for plantation with deep drainage. In the later case, emission from this source is too high to be compensated by the sequestration by oil palm crops. The utilization of peat shrub rather than peat forest for plantation extensification reduces emissions from plant biomass and peat burning, but unlikely changes the peat decomposition component.

Avoided deforestation is the most effective approach of emission reduction from peatland. Once the peat forest is converted, the CO₂ emission escalates and the management systems required by the subsequent land uses determine the emission rates. Other emission reduction strategies from peatland include utilization of degraded shrub land, selection of crops requiring shallow or no drainage, and/or planting of plantation crops with high carbon accumulation capacity. Other management techniques such as fertilization or use of ameliorant such as clay, to some degree also affect the rate of emissions.

The calculation presented in this paper was based on assumptions derived from literature studies. Literature varies in their findings and they may be site specific. Thus, error rate of 40 to 60% is suggested for the emission level. This paper, however, emphasizes more on the approach of estimation rather than the absolute value per se. The values presented are hypothetical, but the relative magnitude could be used as a basis for development of land use and management strategies.

REFERENCES


