IMPROVEMENTS TO GLOBIOM FOR MODELLING OF BIOFUELS INDIRECT LAND USE CHANGE

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Ecofys, IIASA and E4tech are undertaking a study for the European Commission on the indirect land use change impact of conventional and advanced biofuels consumed in the EU. This assessment will be based on the GLOBIOM partial equilibrium model, developed at IIASA.

Recommendations for model improvements were received by the consortium in late 2013 through its stakeholder consultation. Suggestions collected led to a list of potential model improvements that was published in January 2014 and submitted to a new consultation round with stakeholders, the Advisory Committee and the Steering Committee. On the basis of this consultation, a selection of priority improvements to be undertaken under this project was published in March 2014.

The modelling team at IIASA presents in the document below the results of its research on the different selected improvements. These were conducted by reviewing relevant literature, analyzing currently available data through public datasets and input received from stakeholders. The document explains changes performed to the GLOBIOM model on the basis of this research and expected impacts on the results.

A first version of this document was published on 17 September 2014 on the ILUC website. This revised version takes into account comments received on the first document. We are grateful to the readers who provided us their remarks and suggestions and sent us additional data and materials.

Other documents from the project are accessible on the project website: www.globiom-iluc.eu

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Improvement 1: Modelling agricultural residues in GLOBIOM

1. Motivation for improvements

Agricultural residues were so far represented in GLOBIOM only on the supply side, without consideration of competitive uses and sustainability removal threshold. As agricultural residues constitute one of the feedstocks studied in this assessment, it was decided to improve their representation on the supply and demand side to overcome the current loopholes. The representation of agricultural residues will focus in this study on wheat and other cereals straw.[1] Three regions were selected as illustrative case studies. They were chosen taking into account data availability on straw production and uses, a geographic coverage consistent with straw market size,[2] but also contrasting situations with respect to sustainability of additional residue removals, if a 1% biofuels shock from residue is implemented at the national level. The three selected regions are : i) Hungary, as an example of limited availability of straw, as more than half of cereal straw potential is currently being used for feed and animal bedding, which is considered beyond the sustainability removal level, ii) Great Britain as a region with greater availability,[3] but where sustainable potential would be hit if 1% of transportation fuel was supplied from straw biofuels, and iii) Center of France[4] where supply is relatively larger and sustainable potential should not be reached if 1% of French transportation fuel was supplied from straw biofuels. Statistics on characteristics of the different countries are summarized in Table 1 and the contrasting situations of these three regions detailed in Table 2.

2. Methodological approach

Supply of residues: Three different management systems were distinguished to reflect different levels of residue removal: i) no residue removal; ii) sustainable residue removal (around 33-50% depending on the region); iii) high residue removal (greater than sustainable removal). The first and second systems are assumed to have the same biophysical characteristics in terms of crop production (yields, soil organic carbon stocks), but production costs in the second system are higher due to collection of residues.[5] The third management system also has a different collection cost, and modified characteristics for soil organic carbon (SOC).

Because yield can be affected by residue removal, we also consider two different managements for high level of residue removal. The central case assumption is a situation where higher fertilizer rates are applied to compensate for the loss in yield from residue removal. This materializes by an extra production cost and additional fertilizer emissions (not accounted as indirect land use emission). An alternative case will be tested where fertilizer use is not changed, which means residue removal will lead

[1] We will represent cereal straw market in GLOBIOM with straw from wheat, barley, oat and rye, which are found to supply most of straw in Europe (Ecofys, 2013).
[2] Typical transportation distances are reported to be below 500 km.
[3] Excluding Northern Island
[4] Defined as NUTS1 regions FR1 (Ile de France) and FR2 (Bassin parisien).
[5] The cost for residue removal or residue incorporation is based on the data Standarddeckungsbeitragskatalog 2008, from the Austrian Ministry of Agriculture. We follow an assumption of 74 EUR/ha for full residues collection with balling.

to yield decrease. Impact of residue removal on yield is occurring through multiple channels, such as change in soil temperature and moisture, nutrient content, soil texture and sensitivity to water and wind erosion (Blanco-Canqui & Lal, 2007; Johnson & Barbour, 2010). For our sensitivity analysis, we use estimates from EPIC simulations on EU data assuming a linear decrease of yield between sustainable removal rate and yield loss observed when 90% of residues are removed. EPIC simulations only capture a part of the drivers cited above and provide effects on yield of around -0.6% on average after 20 years, with first quartile at -4.8% and third quartile at 0% (see Figure 1). Some other authors find greater impacts on some crops but this is highly dependent on soil type (no impact in two types of soil or up to -15% one type for corn stover in Blanco-Canqui & Lal, 2007; around -10% in Wilhelm, Doran & Power, 1986). Our simulations lead in particular to some positive feedback in case of low input system when residues are removed.[6] We also analyzed with the EPIC model the relative change of soil organic carbon associated to straw removal of 90% (Figure 2). At such rate of removal, under full tillage, SOC decreases after 20 years by 8 ± 3% with some significant differences across locations.

Demand of residues: Several sectors are represented in the model that can compete for residues. First, the livestock sector uses straw as bedding, and to a lesser extent as feed. A generic substitute to straw has been represented in the model for bedding, which allows straw to be replaced by some other materials above a certain price.[7] Animal needs are implemented, with requirements based on Scarlat, Martinov & Dallemand (2010): straw use for cattle is 1.5 kg/day/head for 25% of population, sheep is 0.1 kg/day/head, pigs is 0.5 kg/day/head for 12.5% of population adjusted to (Ecofys, 2013) data when available. Straw used as feed can also substitute with other feedstuff in the livestock sector, with some implications on land use. Additional uses are also considered for energy and horticulture (mushrooms, strawberry, vegetables etc.) as well as industry (material use, pulp and paper). These latter uses are rather small at the EU level (5.5%, 4.8% and 1.5%, respectively, of total residue demand) and are considered as fixed in the model. In total cereal straw uses amounts in the EU28 to 63 Mt per year. Sustainable straw potentials after removal of other uses (Table 1 and Table 2) are consistent with estimates from the Biomass Futures project (BIOMASS FUTURES, 2012), as illustrated in Figure 3.[8]

[6] Producing sufficient and timely quantities of crop residues is expected to increase soil organic carbon and overall soil quality. Incorporated crop residues also support recycling of essential nutrients in the soil and, from long-term perspective, improve soil fertility and have positive impact on yields. However, mineralization is a complex process driven by weather, soil mixing efficiency, soil moisture and nutrients available for microorganisms, and also by the ratio of C (mostly lignin) to N and P in incoming litter. Therefore, EPIC provides quite variable results as these major drivers vary in time and space. Most importantly, high quantities of soil-available N and P are used by microorganisms during plant residue decay (immobilization) which may also negatively impact yields in the following year as nutrients are then lacking for plants. These processes are explicitly included in C, N and P routines in EPIC. Repeated and intensive straw ploughing may therefore have negative effects on yields under management with generally low nutrient inputs. Moreover, other processes including leaching, erosion, runoff, or nitrification/denitrification determine fate of crop residue nutrients and introduce variability into our results.

[7] We currently assume substitute material for bedding available at 22-32 Euro per m^3 of wood chip (0.42 m^3/t wood chip) depending on country according to the Finnish Forest Research Institute (Asikainen, Liiri, Peltola, Karjalainen & Laitila, 2008). One ton of straw requires 1.5 tonne of wood chip in the substitution due to different absorption rate.

[8] Since BIOMASS FUTURES (2012) uses a similar approach to estimate straw demand and sustainable straw potential for bioenergy production after consideration of competitive uses, results are consistent with our data for

Regional markets: Straw is usually not being traded on long distance (usually transported within 500 km maximum). Therefore, we base our representation of local markets on NUTS1 region supply as a general rule, because their size correspond approximately to this order of magnitude. However, for regions where NUTS1 are of relatively smaller size (Germany, UK, the Netherlands), larger units were considered (for instance Great Britain as a whole for the UK). For regions with straw deficit (Netherlands), import needs were added to the demand of neighbour countries.

3. Implication for model results

With this representation of cereals residues, it will be possible to look at land use implications of increasing straw removal. A shock on straw demand at the NUTS1 level will lead in cereal production cost increase in each of the three countries of focus. Residue price will increase, but will be capped by the price of the substitution materials. Cereal, as a join product with straw, will be affected by the extra demand for residues. Primarily on its price, related to the change in price of residues. But also through production level, as food and feed demand reacts to prices. Beyond the sustainability threshold, soil organic carbon stock will also be impacted. Effect on cereal yields will also looked at in the case of a sensitivity analysis, for cases where the suitability threshold is reached. The focus on different regions will help to understand the regional nature of results for straw removal due to the limited extent of trade for this material.

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most European countries (see Figure 3). Main differences can be observed for Germany, Hungary and Poland but can be explained by the use of updated data on animal bedding use from ECOFYS (2013) report whereas data from Scarlat et al. (2010) has been used in BIOMASS FUTURES. Overall, our sustainable straw potential is lightly below the estimates in BIOMASS FUTURES for Europe.

Table 1. Cereals straw balance in 2000 for EU Member States, and impact of a 1% demand shock of bioenergy from straw (1000 tonnes)

Country	Demand	Technical Potential	Sustainable Potential (40%)	Supply - demand	Gap compared to sustainable straw available	Gap after 1% national supply	1% national demand straw req	Share wheat
AT	382	2,535	1,126	744	66%	25%	461	50%
BE	556	1,476	656	100	15%	-73%	577	78%
BG	289	4,741	2,107	1,818	86%	79%	148	82%
CZ	228	5,158	2,292	2,064	90%	73%	399	64%
DE	6,010	32,718	12,360	6,350	51%	23%	3,489	51%
DK	3,635	4,799	2,133	-1,502	-70%	-85%	305	47%
EE	46	638	284	238	84%	65%	54	24%
ES	2,949	16,823	7,477	4,528	61%	28%	2,413	41%
FI	179	3,818	1,697	1,518	89%	72%	298	13%
FR	11,382	36,756	20,420	9,038	44%	30%	2,945	76%
GR	795	2,617	1,163	368	32%	-9%	476	84%
HU	3,522	5,201	1,907	-1,615	-85%	-100%	300	74%
IE	1,191	1,702	756	-435	-58%	-99%	314	31%
IT	1,664	9,094	4,042	2,378	59%	-7%	2,653	83%
LT	148	1,952	868	720	83%	73%	84	49%
LU	30	77	34	4	12%	-446%	156	26%
LV	64	958	426	362	85%	67%	75	46%
NL	1,311	1,087	483	-828	-171%	-336%	797	71%
PL	18,427	24,179	10,746	-7,681	-71%	-79%	758	40%
PT	396	563	250	-146	-58%	-229%	427	66%
RO	1,240	7,595	3,376	2,136	63%	55%	287	79%
SE	289	4,699	2,089	1,800	86%	62%	505	40%
SI	77	198	88	11	13%	-108%	106	76%
SK	112	2,138	950	838	88%	76%	119	65%
UK	7,740	17,698	7,866	1,230	16%	-21%	2,917	66%

Not reported: Malta, Cyprus, Croatia. Demand data for year 2000 were used applying Scarlat et al. (2010) coefficients on livestock number, or when Ecofys data for recent years were available, by applying

Table 2. Selected three case study for the marginal 1% shock at national level (1000 tonnes)

Country	Demand	Technical Potential	Sustainable Potential (40%)	Supply - demand	Gap compared to sustainable straw available	Gap after 1% national supply	1% national demand straw req	Share wheat
Centre France*	3,103	20,253	11,252	8,149	72%	46%	2,945	80%
Great Britain	7,034	17,210	7,649	615	8%	-30%	2,917	66%
Hungary	3,522	5,201	1,907	-1,615	-85%	-100%	300	74%

* This value excludes imports demand from the Benelux.

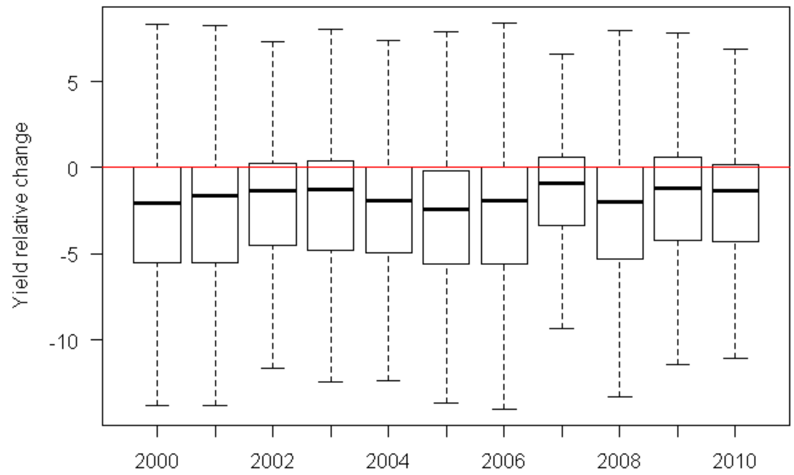

Figure 1. Crop yield relative change (%) in the EU when removing 90% of residues compared to 40%. Ten representative years are shown with their representative climate, after 20 years of removal. Estimates are sourced from the EPIC crop model simulations in all cropland location in the EU. Boxes indicate the first and third quartile of values and whiskers the 5%-95% range.

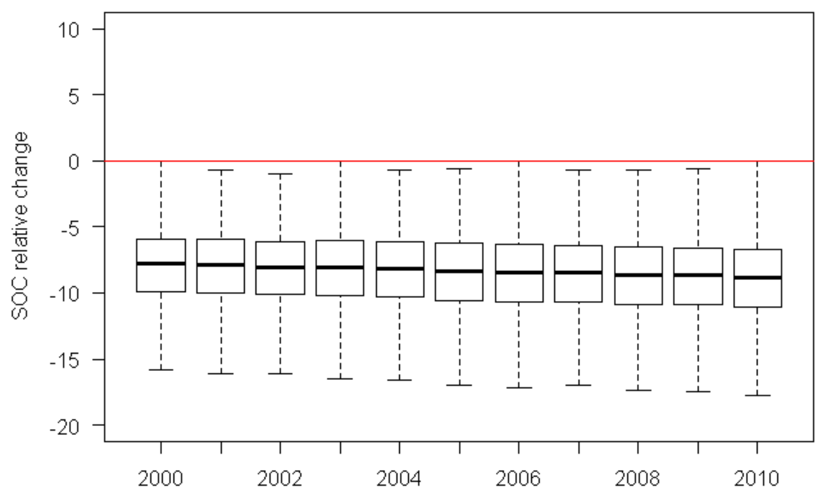

Figure 2. Soil organic carbon (SOC) relative change (%)in the EU when removing 90% of residues compared to 40%. Ten representative years are shown with their representative climate, after 20 years of removal. Estimates are sourced from the EPIC crop model simulations in all cropland location in the EU. Boxes indicate the first and third quartile of values and whiskers the 5%-95% range.

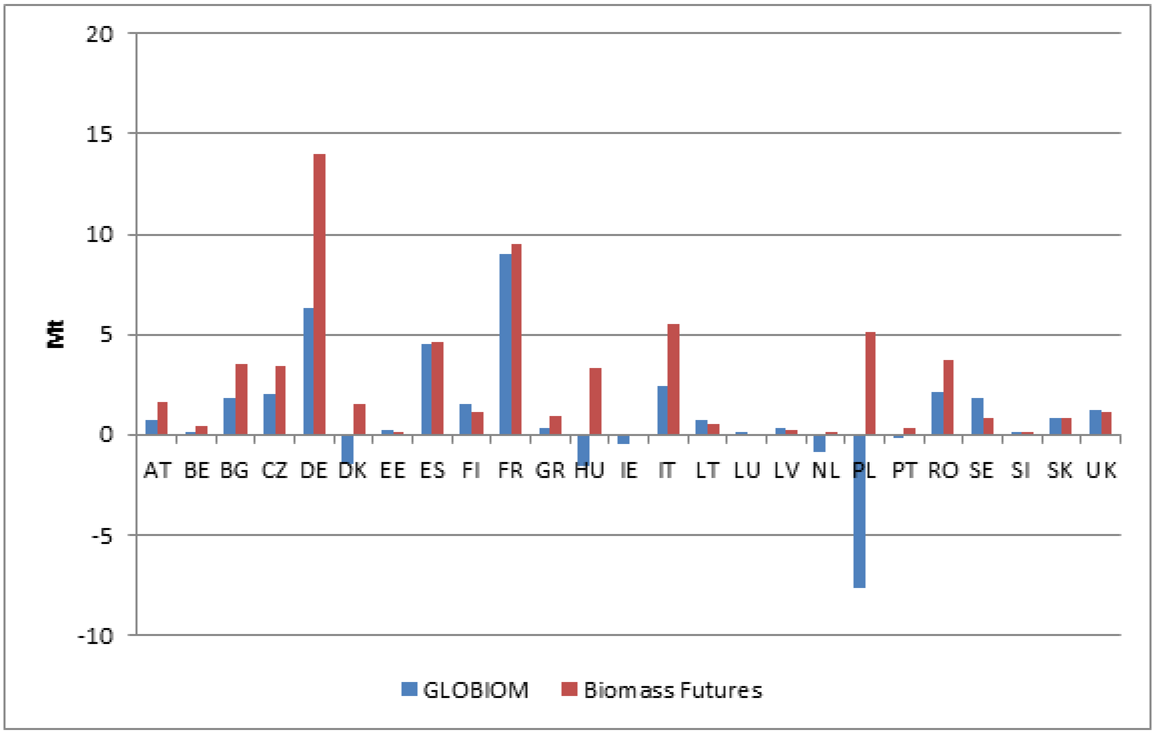

Figure 3. Comparison of straw potential for bioenergy use with data from BIOMASS FUTURES (2012). BIOMASS FUTURES data has been converted from ktoe to Mt wet matter using a LHV of 18 GJ/t and a dry matter content of 85%. IIASA data corresponds to the year 2000 while BIOMASS FUTURES refers to 2004.

Improvement 4 & 5: Representing carbon stocks in annual and perennial crops

1. Motivation for improvements

Land use change drives GHG emissions in particular due to changes in carbon stocks from different biomes. So far, in GLOBIOM, carbon stocks were represented for forest, following statistics from the Forest Resource Assessment 2010 (FAO, 2010) and for grassland, and other natural vegetation, based on the Ruesch & Gibbs (2008) database. However, cropland was not covered. This could lead to some bias in the emissions associated with cropland expansion because some carbon can be sequestered in crops during the harvest cycle, and this for several years in the case of perennial crops.

2. Methodological approach

Different sources were applied to collect carbon stock values associated to each crops. In the case of annual crops, we used the EPIC crop model information, directly related to the management of crop in each simulation unit. EPIC provides crop yield but also dry matter living biomass produced per ha. The IPCC (2006) default value of 0.47 tonne C per tonne dry matter biomass recommended for herbaceous biomass was applied. The carbon stock of annual crops was then multiplied by the fraction of the year during which crops are grown and divided by 2 on the basis of the assumption of a linear growth. In the case of semi-perennials or perennial crops (sugar cane, miscanthus), the crop calendar was applied over the number of years of the plantation cycle. Results for EU crops are provided in Table 3.

A particular case is palm plantations that is modelled as a crop activity in the model but contains carbon stocks that are those of tree plantations. Carbon stock default value for palm oil tree from IPCC (2006) is 68 tonnes C per ha for a mature plantation (136 tonnes dry matter above ground biomass x 0.5 tonne carbon per dry matter tonne as for woody biomass; see IPCC AFOLU Guidelines Chap 5, Table 5.3). However, a typical rotation period for palm oil is 25 years and palm trees are continuously growing on this time period. Therefore, the IPCC value needs to be corrected to account for the growing stock. Khasanah et al. (2012) consider based on several site studies an average of 40 tC/ha on the life-cycle on a plantation, using growth profiles that are consistent with IPCC values for a mature plantation. Therefore, the estimate of 40 tC/ha appears appropriate for above biomass of palm plantations in GLOBIOM. For calculation of below-ground biomass, we also rely on IPCC below to above biomass ration of 0.2 (subtropical humid forest; above biomass lower to 125 t dry matter per ha).

3. Implication for model results

Accounting for cropland carbon stocks should lead to a more comprehensive calculation of CO_2 emissions resulting for land use change, when cropland expands in another land use type or is converted to another use.

Table 3. Above- and below-ground average carbon stock in living biomass for GLOBIOM crops, and annualized stock values. Crops stocks are aggregated at the global level based on their location in 2000.

Crop	Carbon stock at harvest (tC/ha)	Annualized carbon stock in living biomass (tC/ha)
Barley	5.8	2.9
Dry beans	3.4	1.7
Cassava	2.4	1.2
Chick peas	2.5	1.3
Maize	9.6	4.8
Cotton	6.3	3.1
Groundnuts	8.6	4.3
Millet	4.2	2.1
Potato	3.3	1.7
Rapeseed	2.5	1.3
Rice	8.3	4.1
Soybean	6.3	3.1
Sorghum	3.5	1.7
Sunflower	6.4	3.2
Sweet potato	3.1	1.6
Wheat	5.6	2.8
Flax*	3.0	1.5
Peas*	3.2	1.6
Sugar beet*	5.5	2.8
Fallow*	4.2	2.1
Fodder*	3.8	1.9
Corn silage*	2.4	1.2
Oats*	8.5	4.3
Rye*	4.1	2.0
Grassy crop**	10.0	10.0
Sugar cane***	13.4	13.4
Oil palm	--	48.0

* for EU only

** On the basis of miscanthus

*** Carbon stock after one year, harvest after two years.

Improvement 7: Emission factors for oil palm on peat

1. Motivation for improvement

Past studies on indirect land use change (ILUC) have found biodiesel consumption to impact palm oil production, directly or indirectly. Expansion of palm plantations in Indonesia and Malaysia, which represents 80% of global production (FAO, 2014), has occurred for a significant share on tropical peat soils (Gunarso, Hartoyo, Agus, & Killeen, 2013). As a consequence of soil being drained, the peat starts to slowly decompose and can emit greenhouse gas (GHG) for several decades.[9] Although the number of studies that estimate these peat emissions has increased tremendously during the last decade, the scientific community has not yet reached a consensus on an appropriate range of emission factors. For instance, revision of IPCC guidelines for wetlands has been hotly debated (IPCC, 2013).

So far, GLOBIOM could only take into account this type of emission by mapping cropland area to the organic soil emissions reported in FAOSTAT at the national level. The objective of this improvement is to define, on the basis of the existing literature, a more specific range of emission factors for peat drainage[10] for the ILUC simulations. This range of values will then be used in the Monte-Carlo simulation, i.e. an iterative approach to take account of a range of plausible emission values. Moreover, this section also aims at providing an overview of the most prominent drivers of GHG emissions from peat and explains the most critical methodological issues that may explain the scientific disagreement observed between the various author groups.

2. Methodological approach

Our analysis of potential emission factors builds here upon a wide examination of past literature.

A number of determinants were identified that drive the pace and the magnitude of GHG emissions from peat:

- The **level of the water table (drainage depth)** directly determines peat decomposition rate. Several scholars provide estimates of GHG emissions per additional centimeter of drained soil (Hirano et al. 2012; Hooijer et al. 2006; Wösten et al. 1997).
- **Natural respiration variability and timing of measurement:** Both intra-annual changes (e.g. temperature and rainfall distribution over the year) and inter-annual changes, such as the el Niño phenomenon can explain a significant part of the variability observed in measurement. Additionally, peat respiration curves show a tendency to peak over the 5-10 years after drainage followed by a flattening of the emission curve (Page et al., 2011), which needs to be taken into account to provide reasonable emission factors.

[9] CO_2 is by far the most prominent GHG, accounting for about 98% of all peat-related GHG's (Hergoualc'h and Verchot 2013; Schrier-Uijl et al. 2013). CH_4 and N_2O, the later mainly upon application of mineral fertilizers, constitute the remainder of the total GHG emissions. In the present literature review we thus focus on the role of CO_2.

[10] Due to methodological uncertainties, immediate emissions from peat fires and emissions from peat drainage in forests adjacent to plantations will not be considered in the ILUC assessment.

- **Current and past land use and management:** land use and land management affect the level of peat oxidation and thus the measured emission flow. For instance, fertilization practices stimulates microbial soil activity and can increase peat emissions; furthermore, different types of land use imply different drainage depths (Dariah et al., 2013).
- **Peat bulk density (BD) and the fraction of carbon in soil** influence peat decomposition rates. BD values vary throughout the soil profile and need to be sampled with care as they feed directly in the formula for emissions in the case of studies based on measurement of the soil subsidence (Melling and Henson 2011; see Box 1).
- The **measurement method** used to estimate fluxes of GHG from peat to the atmosphere, namely measurements of soil subsidence, of direct flux measurement through closed chambers, and measurements by Eddy Covariance techniques (see Box 1).

Table 4 and Table 5 provide an overview of 12 studies based on subsidence and closed chambers, respectively, and lists some of the determinants mentioned above. No Eddy covariance studies were found for oil palm plantations.

Subsidence studies find the highest potential emissions, due to the full accounting of the emission cycle along the exploitation process of plantation. The method strength relies on the explicit representation of peat oxidation process but due to the long period of study required, only a few estimates are available. Estimates critically depend on the subsidence rate. Hooijer et al. (2012) find the highest estimates as they also account for the initial subsidence in the few years following the drainage, whereas other studies look at emission fluxes for a period after 5 years of drainage.

Close chambers studies are more numerous but the range of their results is highly variables. Earlier studies were flawed by methodological problems, such as interference of root respiration,[11] too short periods of measurements and bias due to the time of measurement in the day. Figure 4 shows that closed chamber estimates tend to increase over the past recent years and the most extreme points corresponds to non-peer-reviewed results (Melling et al., 2007; Agus et al., 2010; Comeau et al. 2013). The lowest published value is from Dariah et al. (2013) with measurements at 34.1 and 38.2 MtCO_2-eq ha^{-1} yr^{-1} and the highest to Husnain et al. (2014) with 66 MtCO_2-eq ha^{-1} yr^{-1} and Jauhiainen et al. (2012) with 80 MtCO_2-eq ha^{-1} yr^{-1}. This last estimate are however source from several acacia sites, and authors disagree on whether such flux chamber measurements are directly transposable to the case of palm oil plantations.[12]

[11] Trees on the plantation site emits CO_2 through root respiration (autotrophic respiration) which is also captured by closed chambers

[12] The question whether acacia and palm oil should be considered similar is still unresolved. IPCC (2013) rejected comments from the US government to consider acacia and palm oil plantations equivalent pointing four reasons that could justify differences: i) shorter rotation time of 6 years versus 25 years leading to higher soil disturbance, ii) difference in fertilization and nitrogen cycle, iii) larger depth of drainage for acacia iv) different regions of plantations. However, they also acknowledge that studies could have reported different values for the two types of plantations due to some different sites being looked at. Some more recent studies (Couwenberg and Hooijer, 2013 on subsidence; Husnain et al., 2014 on closed chambers) suggest that differences might not be as high as for the

The three to four research groups publishing actively on peat land emissions also authored a number of literature reviews (Table 6). Their usually reported estimates and recommendations vary. Page et al. (2011) repeatedly find high emission rates and recommended a value of 95 t CO_2-eq ha^{-1} yr^{-1} (based on Hooijer et al. 2012) while the group around Agus usually reports emissions of 43 t CO_2-eq ha^{-1} yr^{-1} (Agus et al., 2013; A. Hooijer et al., 2010, 2012). IPCC (2013) chose a Tier 1 emission factor of 40 tCO_2-eq ha^{-1} yr^{-1}.

Sources of uncertainty are too large to lead to a narrow estimate of peat emissions in South-East Asia given that emission estimates. To derive our final range of estimate, we proceed in two steps:

- **filtering of studies:** all studies analysed in this review are not equal in terms of level of details, robustness of the methodology and validation of the results. To improve the quality of our reference values, we decide to consider as relevant only the values produced by studies respecting two criterias: i) peer-reviewed and ii) in the case of closed chambers, we only considerer studies separating autotrophic (root respiration) and heterotrophic (peat oxidation) calculations, a bias that can play a significant role around trees (Dariah et al., 2013). As a consequence, three field studies are removed from our sample: Melling et al. (2007), Agus et al. (2010), Comeau et al. (2013).[13] This particular leads to removal on the lowest and highest values in our closed chamber range for palm oil. In addition, although we kept in the sample the measurements on acacia, we displayed them separately due to the on-going debate about differentiated impact of peat drainage for palm oil or for acacia.[14]

- **distribution of emissions:** if we follow the subsidence method, emissions depends on different uncertain multiplicative drivers, among which oxidation rate, peat bulk density and subsidence rate (related to water table level). There is no large scale dataset on the distribution of these factors over the regions of interest for our study. If we assume such values are symmetrically distributed and independent, the resulting distribution should be log-normal shaped.[15] This profile is confirmed by observation with flux chambers (see for instance records from Dariah et al. (2013). Based on some distribution on oxidation rate in the range 40-92% (Page et al., 2011; Hooijer et al., 2012), 0.06-0.12 g $cm^{-3}$for peat bulk density at 55% C (Jauhiainen et al., 2012) and a water table of 0.6-0.85 m (Page et al., 2011), we can reproduce a distribution profile consistent with the literature. The mean of the distribution is 61 ± 22 tCO_2 ha^{-1} yr^{-1}. The median value is 58 tCO_2 ha^{-1} yr^{-1} and the confidence interval at 95% in the range 27--112 tCO_2 ha^{-1} yr^{-1}. The first quartile of the distribution is at 44 tCO_2 ha^{-1} yr^{-1}, which is in the magnitude of the Tier 1 value of IPCC (2013). The third quartile is 74 tCO_2 ha^{-1} yr^{-1}, which is

currently proposed Tier 1 emission factor from IPCC (11 tC ha^{-1} yr^{-1} for palm oil and 20 tC ha^{-1} yr^{-1} for acacia) and can be of similar magnitude for a same site (Husnain et al., 2014).

[13] Some of the sources above appear in particular very little documented. Melling et al. (2007) is only three pages of explanations, not peer-reviewed; Agus et al. (2010) is the same level of details, with a very succinct results section and no peer-review. Comeau et al. (2013) is more developed, but do not distinguish the effect of root respiration (autotrophic respiration).

[14] See footnote 4 above.

[15] A log normal distribution is the distribution of a random variable whose log value is normally distributed. It is typically characterized by a longer right tail and a mean value higher than the median value. Log-normal distributions are usually observed when evenly distributed random variables are multiplied together. For illustration of the role of log-normal distribution in science, see (Limpert, Stahel, & Abbt, 2001)

above most closed chamber measurements, but below measurements on acacia plantations (Jauhiainen et al., 2012).

Limitations to this approach: Average range of subsidence assumed here is 5cm/yr, with a confidence interval of 3.4cm yr^{-1} to 7 cm yr^{-1}. This is in line with most records from subsidence at steady state, but this does not account for the emission peak of the first five years described by Hooijer et al. (2012) on the observation of an acacia plantation. However, we did not find any study quantifying the effect of such a peak on a palm plantation. The peak effect can be partly represented through high bounds of peat bulk density, typical of higher layer of peat, and the higher values of our subsidence rate, but might be underestimated compared to Page et al. (2011) for instance. More comprehensive information on subsidence rate distribution could help overcome this caveat but are not yet available. Another limitation comes from the assumption that some variables are independent, such as oxidation rate and age of the plantation (reflected through subsidence rate). The most recent publications suggest that the oxidation rate should increase with the age of plantation and compensate the decrease of subsidence rate with the plantation aging (Hooijer et al. (2012), Couwenberg & Hooijer (2013)). However, information to properly quantify this relation on a systematic basis is not yet available.

As more sites will be monitored over time, we can progressively expect better information on key drivers distributions on oxidation rates, subsidence rate and peat bulk densities. For the time being, **we base our final range on the simple subsidence relation above, which covers the current observations for an average water table level of 0.6-0.85 m with a mean value of 61 ± 22 tCO_2 ha^{-1} yr^{-1} and a 95% confidence interval of 27--112 tCO_2 ha^{-1} yr^{-1}.** As a more comprehensive set of records from the literature will be recorded, this method will be able to evolve and the range be refined.

3. Implication for model results

This range of peatland emission factors will be used with the expansion pattern of improvement 29 to account for impact of palm plantation expansion. These emissions being highly uncertain, they will be clearly identified in the total accounting of land use change emissions.

Box 1. Measurement methods used for peatland emissions

Subsidence: Drainage of peat results in the increase in the oxidation of carbon and the transfer of carbon to the atmosphere. The removal of carbon from peat results in its shrinkage and the increase in its bulk density, thereby resulting in peat subsidence. CO_2 emission estimates may be based on peat subsidence as they are related to one another. Carbon loss is calculated using the formula:

$$C_{loss} = S_t \times DBD_l \times C_{dw}$$

Where S_t is the surface height loss, DBD_1 is the dry peat bulk density and C_{dw} is the volumetric carbon density of peat below the water table (the product of carbon bulk density and carbon concentration in the peat). Estimation of the contribution of the oxidative component to overall subsidence is critical in order to infer soil CO_2 emissions. Assumed or calculated oxidation rate is a source of uncertainty across studies with values in the range 40-90% (Wösten et al. 1997; Hooijer et al. 2012; Couwenberg et al. 2009). The subsidence-based CO_2 estimates are cost-efficient, allow for high spatial resolution in the sampling process, and emissions from peat (heterotrophic emissions) can clearly be distinguished through this method from vegetation emissions (autotrophic emissions). The technique yields results which are comparable to techniques that measure emissions directly (Page et al., 2011). Yet, emission estimates are limited to CO_2 (no CH_4 or N_2O) and they critically depend on the estimation of the oxidative fraction of peat subsidence which is subject to a high uncertainty (Dariah et al. 2013).

Closed chambers: they provide direct measurements of gas fluxes from the soil at discretionary spatial and temporal resolution, the results of which can be up-scaled to obtain emission factors for given site conditions. Rigorous measurement scheme allows for reliable measurements of heterotrophic soil emissions. Homogenous experiment set-up (e.g. chamber location in the micro-relieve and varying chamber sizes) is imperative in order to obtain reliable results. The advantage of this method is the direct measurement of emissions, and the number of samples collected so far. However, a certain number of technical challenge weaken the reliability of measurements, due to the high variability of results, depending on the location and the moment in the year or in the day where experiment are conducted, and the risk of measurement bias related to the distance to root of planted trees that can create interference between heterotrophic respiration from peat oxidation and autotrophic respiration from roots (Page et al., 2011).

Eddy Covariance (EC): method to measure gas fluxes on towers reaching above the top of the vegetation cover. It is suitable to capture the total GHG balance of larger sites with trees, but it is limited by high costs, low portability and low spatial resolution. EC studies on peat were presented by Hirano et al. (2007; 2012) but not for oil palm, thus we did not consider them further in this review[16].

[16] Similar observations were made on EC by IPCC (2013).

Table 4 : Summary of available data from studies based on the subsidence method on plantations on peat (Source: authors' compilation).

Number	Study (year)	Study peer review-ed	Affiliation / funder	Location		Average observed subsidence [cm/year]	Determinants					Mean estimated emission factor [t CO_{2-eq} ha^{-1} yr^{-1}]	Associated range[17] [t CO_2 ha^{-1} yr^{-1}]
				Land use	Location		Drainage depth [m below ground surface]	Time after drainage [years]	Duration of estimation [months]	Observed peat bulk density [g cm^3]	Oxidation rate observed or applied		
1	Wösten et al. (1997)	Yes	Wageningen / Malaysian Ministry of Agriculture	Oil palm	Sarawak, Malaysia	4.6[18]	0.7	14-28	275	0.1	60%	**61[19]**	30-91[20]
2	Hooijer et al. (2012)	Yes	Singapore-Delft Water Alliance	Oil palm	Sumatra, Indonesia	4.3-6.5[21]	0.5 – 1.06[22]	14 - 19[23]	24	0.07–0.09[24]	92%[25]	**109[26]**	47-119[27]
3	Couwenberg & Hooijer (2013)[28]	Yes	Singapore-Delft Water Alliance	Oil palm	Sumatra, Indonesia	3.2-4.4[29]	0.4 – 0.9	5 – 20	36	0.08 – 0.13[30]	~80%[31]	**62.4**	51-75[32]

[17] Information based on interpretation by authors of this note under assumptions below. Not provided by authors of the studies.
[18] The study reports the profile of subsidence for a long period with 4.6 cm/year for 14 to 28 years of age and 2 cm/year beyond. We apply here the subsidence rate of the earlier period.
[19] Assume 60% decomposition rate and subsidence rate of 4.6 cm/year. The initial published value in Wösten et al. (1997) is 26.5 tons CO_2-eq ha^{-1} yr^{-1}, on the basis of subsidence of 2 cm/year but correspond to a more aged plantation (>28 years).
[20] Replicating the sensitivity analysis of the authors on peat bulk density.
[21] Value observed for plantation older than 5 years old: 5.4 cm/yr on average with standard deviation of 1.1 cm/yr. Very large subsidence rate was observed in the case of acacia in the 5 years after drainage and used for the calculation on palm oil (142 cm in 5 years).
[22] 73 cm drainage measured in particularly wet year – usually WT is lower
[23] Emissions from first years of plantation are also accounted for on the basis of acacia measurements.
[24] Assumption of homogenous BD and carbon content over soil profile
[25] Oxidation rate is here directly inferred from bulk density measurements and subsidence rate, assuming steady state in the subsidence process.
[26] Estimate is an annualized value over 25 years rotation time – taking into account the first five years of a very high emission level (178 tCO_2-eq/yr) and then 73 tCO_2-eq/yr. The high initial subsidence rate was measured on acacia plantation. Assume 70 – 92% decomposition rate.
[27] Using the subsidence range on a period of 25 years, with the 5 first year collapse of peat observed in acacia plantation and without it.
[28] The study looks at three sites, in same provinces as in Hooijer et al. (2012). Calculation methods differ and the study only looks here at emissions after peat consolidation (over 5 years).
[29] Site of young plantation showed subsidence of 3.7 ± 0.5 cm/yr and old plantation 3.9 ± 0.5 cm/yr.
[30] Only lower layer value is used in the steady state calculation.
[31] Due to the carbon loss difference method chosen, oxidation rate is directly derived from peat density and subsidence measurements.
[32] Using the confidence interval on subsidence rate.

Table 5 : Summary of available data from peat carbon emission studies on plantations on peat.

Study number	Study (year)	Study peer review-ed	Affiliation / funder	Contextual information			Determinants				Mean estimated emission [t CO_2-eq ha^{-1} yr^{-1}]	Stdev. or range of estimated emission [t CO_2-eq ha^{-1} yr^{-1}]
				Land use	Location	Number of sites	Drainage depth [m below ground surface]	Time after drainage [years]	Duration of estimation [months]	Separation auto- and hetero-trophic respiration		
4	Melling et al. (2005)	Yes	Tropical Peat Research Lab. / Malaysian ministry of Science	Oil palm	Sarawak, Malaysia	1	0.6 (variable)	7	12[33]	No	**60.6**	**15-107**
5	Melling et al. (2007)	No	Tropical Peat Research Lab. / Malaysian ministry of Science	Oil palm	Sarawak, Malaysia	1	NA	5	12	Yes	**33.6/40.1**[34]	**NA**
6	Agus et al. (2010)	No	Indonesian Soil Research Institute	Oil palm	Sumatra, Indonesia	3	0.7 – 1.5	1-10	2[35]	Yes	**19.5**	**±13.2**
7	Jauhiainen et al. (2012)	Yes	SDWA / Grant of Academy of Finland	Acacia[36]	Sumatra, Indonesia	8	0.45 – 1.39	7	24	Yes	**80**[37]	**±15**[38]
8	Dariah et al. (2013)	Yes	Indonesian Soil Research Institute / EU FP7 program	Oil palm	Sumatra, Indonesia	2	0.52 – 0.58	8	10	Yes	**34.1/38.2**[39]	**±9.5/15.9**
9	Marwanto	Yes	Indonesian Soil	Oil palm	Sumatra,	1	0.59 –	15	12	Yes[40]	**46**	**±30**

[33] Short daily measurement period (two hours) – unclear if measurements are representative
[34] Melling and Henson (2011) report 33.6 $MtCO_2$ which corresponds to microbial respiration, Marwanto & Agus (2013) also report other soil emissions not associated to roots.
[35] Measurement period of 2 months only.
[36] This study is looking at acacia palm but is retained here because it has been largely cited and also discusses application of findings to palm oil plantations.
[37] The authors of the study reduce the daytime measurement of 94 tCO_2-eq ha^{-1} yr^{-1} by 14.5% to account for night temperature correction.
[38] After applying the same correction same correction on standard deviation as for the mean.
[39] Lower value for a plantation aged of 15 years, higher value for a six-year-old plantation.
[40] No explicit distinction is performed but measurements were performed sufficiently far from the palm tree according to authors.

Study number	Study (year)	Study peer review-ed	Affiliation / funder	Contextual information			Determinants				Mean estimated emission [t CO_2-eq ha^{-1} yr^{-1}]	Stdev. or range of estimated emission [t CO_2-eq ha^{-1} yr^{-1}]
				Land use	Location	Number of sites	Drainage depth [m below ground surface]	Time after drainage [years]	Duration of estimation [months]	Separation auto- and hetero-trophic respiration		
	& Agus (2013)		Research Institute / EU FP7 program		Indonesia		1.27					
10	Comeau et al. (2013)	No	CIFOR / Australia, Norway and EU FP7 program	Oil palm	Sumatra, Indonesia	1	0.65 – 1.05	10	9	No	**104**	**±4**
11	Melling et al. (2014)	Yes	Tropical Peat Research Lab. / Malaysian ministry of Science	Oil palm	Sarawak, Malaysia	3[41]	0.56 – 0.66	1-7	24	Yes	**60.1**[42]	**±3**
12	Husnain et al. (2014)[43]	Yes	Indonesian Soil Research Institute / EU FP7 program	Oil palm	Sumatra, Indonesia	1	0.2 – 1.4	7	7 - 13	Yes	**66**	**±25**

Source: authors' compilation

[41] No separation of auto- and heterotrophic emissions
[42] Median value for a 5 year-old palm plantation. Authors report 54 and 68 tCO_2-eq ha^{-1} yr^{-1} for a one year and seven-year old plantation, respectively.
[43] The study uses results from two other studies already listed here: Marwanto & Agus (2013) and Dariah et al. (2013). To avoid double-counting we only report here the specific site added by the paper in the Riau province.

Table 6 : Overview of reviews and meta-studies on peatland emissions. (Source: authors' compilation)

Study (year)	Peer-reviewed study	Affiliation / funder	Number of studies	Common assumption	Final range of estimates [t CO_2-eq ha^{-1} yr^{-1}]	Recommended emission factor [t CO_2-eq ha^{-1} yr^{-1}]	Comment
Verwer, Meer, and Nabuurs (2008)		Alterra, Wageningen	--	• Take 60-80 cm drainage		10 per 10cm drainage depth	• Quantitative estimates based mostly on Hooijer et al. (2006)
Uryu et al. (2008)		WWF Indonesia	2	• Average drainage depth of 53 cm	5 - 165	85	• Estimated values (drainage depth, emission factors) based on studies of Melling and the Hokkaido University • Point out the large variations of drainage depth as a function of the weather (e.g. El Niño)
Couwenberg (2009b)	x	Univ. Greifswald / Wetlands International	--	• Per 10cm of drainage depth • For 50 – 100cm drainage depth, 40% of subsidence caused by oxidation	--	≥ 9 per 10cm drainage depth	• Based on Couwenberg et al. (2009a)
Hooijer et al. (2010)	x	Deltares, SDWA	7 studies reporting water table depth	• Drainage depth 0.95 m (0.80 – 1.10 m) • 0.91 t CO_2-eq ha^{-1} yr^{-1} per cm of drainage	73 - 100	86	• Relation of WT-depth and emissions based on Hooijer et al. (2006)
Page et al. (2011)		Univ. Leicester / International council of Clean Transportation (ICCT)	12	• Drainage depth 0.6 – 0.85 m	54 - 115	95	• Recommended value based on Hooijer et al. (2012) • Exclusion of some studies for methodological flaws • Annualized value over 30 years rotation time
Hergoualc'h and Verchot (2011)	x	CIFOR/ Grants from Australia and Finland	11 (2 for oil palm)	• Drainage depth 0.60 m (0.55 – 0.65)	24.1 – 44.1	34.1	• Meta-model based on sample of studies (input-output method) • Includes CH4 and N2O (ca. 2% of total emissions)
Melling and Henson (2011)	x	Tropical Peat Research Laboratory Unit, Malaysia	19 (8)	--	33.6—89.8	--	• Review also CH_4 and N_2O
Agus et al. (2013)		Indonesian Soil Research Institute / Round Table on Sustainable Palm Oil (RSPO)	14	• Drainage depth 0.5 – 0.7 m	20--95	43	• Recommended value based on recalculation by Agus et al. (2013) of results by Hooijer et al., (2010) and proposed correction factor for root-related respiration by Jauhiainen et al. (2012).--> Approach challenged by Schrier-Uijl and Anshari (2013)
Hergoualc'h and Verchot (2013)	x	CIFOR / Grants from Australia, Norway and EU FP7 program	28	0.65 +/- 0.05 m	35.2-50	45.1	• Meta-model based on sample of studies • Includes CH4 and N2O (ca. 2% of total emissions)

Table 7 : Subsidence parameters selected for our distribution of peatland emission factors and results (Source: authors' calculations)

Parameter	Notation	Unit	Range		Distribution	Source		Comment
			Min	Max		Min	Max	
Water table	WT	cm	60	85	Uniform	Hergoualc'h and Verchot (2013); Page et al. (2011)	Page et al. (2011); Hooijer et al. (2010)	Typical drainage for oil palm cultivation is supposed to be 0.7 m for oil palm and recommended depth is 60-80 cm (Verwer et al. 2008a; Mutert et al. 1999). Lower depth can be observed (Agus et al., 2013 report low bound at 0.5 m) but can also be much higher in industrial plantations (Hooijer et al., 2010). We follow Page et al. (2011) that cover the most common values also found in Table 1-3.
Subsidence / Drainage depth	r	$cm\ yr^{-1}\ cm^{-1}$	0.05	0.09	Uniform	Wösten et al., (1997)	Wösten et al. (1997); Couwenberg et al. (2010)	Wösten et al. (1997) were the first to propose an average coefficient of 0.07 to link water table and subsidence rate. His proposed range is 0.04-0.09, however, his measurement for the low bound correspond to a plantation more than 30 years old. We therefore conserve the symmetry around 0.07. Couwenberg et al. (2010) find a coefficient of 0.09 for the first 50 cm but suggest the correlation could be not applying beyond this depth. Hooijer et al., (2012) examine the relation for an acacia plantation and find a slope of 0.0498 with however an intercept value of 1.5 cm yr^{-1}. For 0.7m drainage, this regression is consistent with the linear relation from Wösten et al. (1997). They note that they could not find a clear relation on the palm plantation with more homogenous subsidence rates.
Peat bulk density	BD	$g\ cm^{-3}$	0.06	0.12	Uniform	Jauhiainen et al. (2012); Couwenberg et al. (2010) ; Couwenberg & Hooijer (2013)	Jauhiainen et al. (2012) ; Hooijer et al. (2010) ; Couwenberg & Hooijer (2013)	Peat bulk density profiles are reported in Hooijer et al. (2012), Couwenberg and Hooijer (2013), decreases significantly along the peat profile, that vary between 0.06 to values up to 0.15 g cm^{-3} for the top 10 cm. Couwenberg et al. (2010) used a density of 0.068 g cm^{-3} for lower peat layers and Jauhiainen et al. (2012) values in the range 0.06-0.12 g cm^{-3}. Couwenberg and Hooijer (2013) observe values around 0.12 g cm^{-3} for the upper 0.5m peat layer and around 0.08 g cm^{-3} for lower layer.
Oxidation rate	Ox	%	40	92	Uniform	Couwenberg et al. (2010) ; Page et al. (2011)	Jauhiainen et al. (2012) ; Hooijer et al. (2012)	Couwenberg et al. (2010) report range in the literature of 35-100% but applies in his calculation a range of 40-60%. Page et al (2011) performs various analysis using 40% and 60% oxidation rate. Jauhiainien et al. (2012) find higher oxidation rate of 80% and Hooijer et al. (2012) report a measure oxidation rate of 92%.
Carbon fraction	Fc	%	50	60	Uniform	Couwenberg et al. (2010) ; Agus et al. (2013)	Page et al. (2011)	Jauhiainen et al. (2012), Hooijer et al. (2012), Couwenberg & Hooijer (2013), Couwenberg et al. (2010) all use a carbon fraction of 55%. Page et al. (2011) use 60%, whereas Agus et al. (2013) note that variation of carbon fraction over the peat profile must be better taken into account. Couwenberg et al. (2010) report some possible slightly lower carbon fraction on peat with average in some samples at 50%.

RESULTS

Parameter	Notation	Unit	**Range (95%)**		**Mean**	Distribution			Comment
			Min	**Max**		25%	50%	75%	
Subsidence rate (=r · WT)	S	$cm\ yr^{-1}$	**3.4**	**7.0**	**5.1**	4.3	5.0	5.8	The range of subsidence obtained covers well values reported by the subsidence literature (see e.g. discussion in Hooijer et al. (2012))
Emission per cm drainage (=100 · r · BD · Ox · Fc · 44/12)	e	$tCO_2\ yr^{-1}\ ha^{-1}\ cm^{-1}$	**0.39**	**1.52**	**0.84**	0.62	0.8	1.02	Emission per cm drainage range encompasses here values from Agus et al. (2013): 0.72, Hooijer et al. (2010): 0.91 and Jauhiainen et al. (2012): 0.71 with an intercept.
Emission factor (= e · WT)	EF	$tCO_2\ yr^{-1}\ ha^{-1}$	**27**	**113**	**61**	44	57	74	Emission factor obtained match well the range from filtered literature as shown in Figure 5.

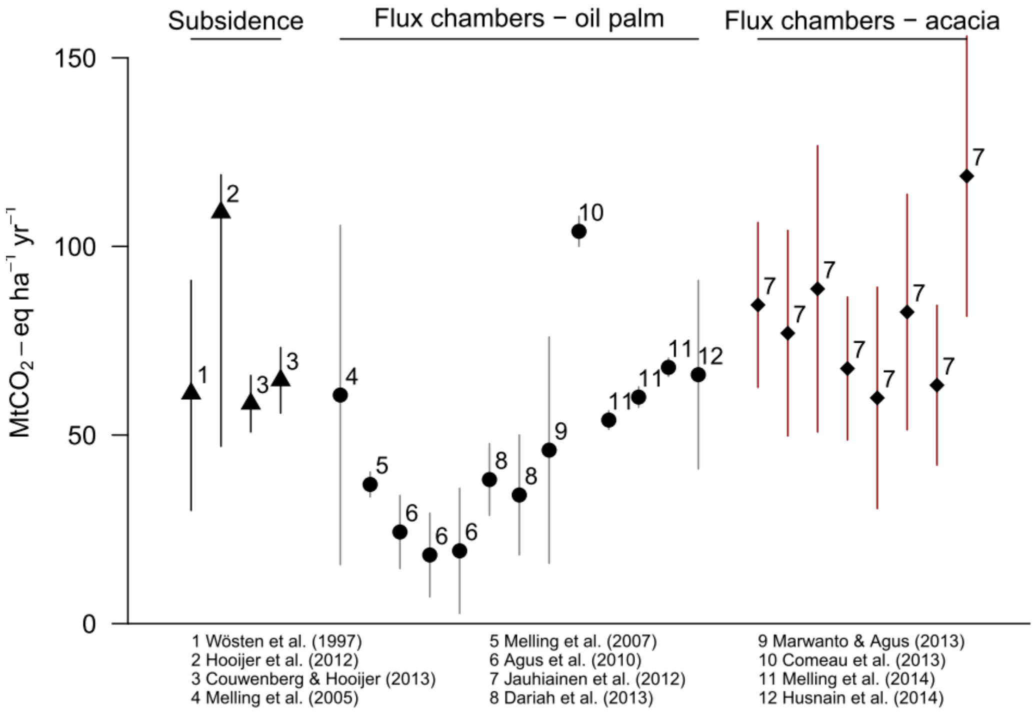

Figure 4 : Distribution of central estimates of studies and range of uncertainty. Values are reported according to Table 4 & Table 5 statistics. For papers analyzing different sites, the different findings were reported separately. For subsidence, the range of uncertainty corresponds to sensitivity analysis on subsidence rate or peat bulk density. For Flux chamber studies, the standard deviation is reported, except for Marwanto & Agus (2013) where only the min and max were available. In each group, results are ordered by year of publication. Acacia values are displayed separately.

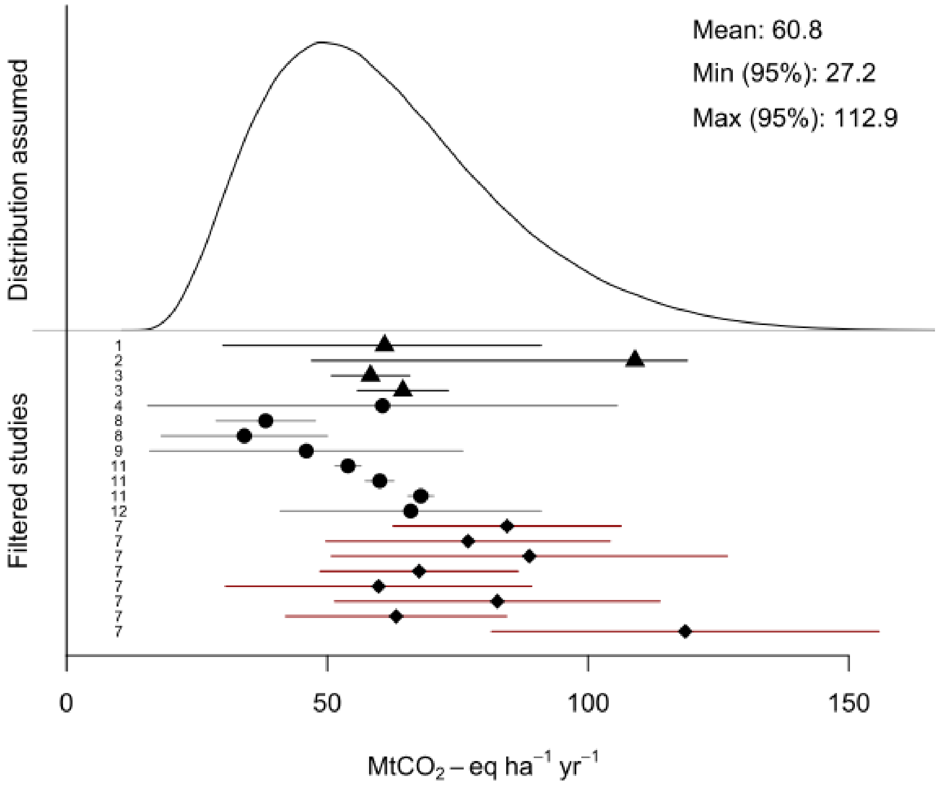

Figure 5 Distribution used for the distribution of emission factors based on simplified subsidence assumptions and comparison with literature values. The upper part of the graph shows the distribution assumed. The lower part features the articles used in Figure 4, after a filtering process. Numbers in the lower part refer to the study names in Figure 4.

Improvement 29: Expansion of oil palm plantations on peat[44]

1. Motivation for Improvement

Palm oil production is a significant source of GHG emissions when new plantations are developed on peat land. Until recently, spatially explicit data on recent development of plantations in Southeast Asia was scarce. As a consequence, future expansion patterns are difficult to predict, and the number of local drivers and extent of policy-driven uncertainty make this dynamics difficult to model. Currently, GLOBIOM does not represent in Indonesia and Malaysia internal transportation costs, which means plantation expansion is only allocated on a crop suitability basis. Therefore, it was decided in the context of this project to ground assumptions on location of palm plantation expansion on the basis of current literature findings. The model will then use different possible allocation within a plausible range as an input in the Monte-Carlo sensitivity runs. Expansion into peat land will then be associated emission factors derived from improvement 7.

2. Methodological approach

We look at the literature findings on two different aspects: first, the estimation of current peat land occupied by palm oil plantation, to get insight into the average share of plantations that expanded into peat land in the past, and the trend of expansion; second, the projection patterns assumed for future expansion of plantations into peat land, also a topic of exploration of some papers.

Historical expansion: We reviewed five studies which assess recent development of palm plantation on peat based on a remote sensing analysis (see Table 8).[45] Gunarso et al. (2013) report that in 2010, palm plantations grown on peat accounted for 1.7 million ha in Indonesia and ca. 721,000 ha in Malaysia, which represents 22% and 18% of the total plantation area, respectively. Miettinen et al. (2012) found 1.3 million ha in Indonesia and ca. 780,000 ha in Malaysia. However, a considerable area of Peninsular Malaysia and East Kalimantan was not included in their analysis due to persistent cloud cover on satellite images, which might partly explain the lower estimations as compared to Gunarso et al. (2013). The third study covering both countries found notably lower numbers with ca. 508,000 ha of converted peat land in Indonesia and ca. 371,000 ha in Malaysia (Koh, Miettinen, Liew, & Ghazoul, 2011). This difference is likely due to the coarser scale satellite imagery applied in their study[46], which did not allow for identifying i) immature palm plantations (<80% canopy cover) and ii) small patches of plantations (<200 ha). Palm expansion after 2002 could therefore not be considered, whereas the annual expansion rate ranged between 8-10% in Indonesia and 3-6% in Malaysia on that period (Gunarso et al., 2013).

[44] Improvement 29 is referenced here (between improvement 7 and 8) due to its direct connection with improvement 7.

[45] Some older assessments also exist that have based their estimation on analysis of palm concession maps and not on remote sensing analysis of current planted areas. This usually led to higher estimate of peat land occupation (for instance, 25% in Indonesia in Hooijer et al., 2006) because the full concession area can be sometimes little developed.[45] This however suggests that future expansion could still drive larger share of expansion into peat land, if all currently attributed concession areas were developed.

[46] For most papers, current distribution of plantations were analysed using high to medium resolution satellite imagery (5-30m resolution). Visual interpretation of satellite images followed by manual delineation of oil palm stands was the most common approach to identify plantation areas (Carlson et al., 2012; Gunarso et al., 2013; Miettinen et al., 2012), sometimes combined with object-oriented digital classification (Omar et al., 2010). Koh et al. (2011) use lower resolution imagery (250m resolution) and applied a different classification algorithm.

Dynamics of expansion: we analyse in details the expansion patterns provided by Gunarso et al. (2013).[47] These show a strong increase in the share of expansion in plantations occurring on peat land in Malaysia over the years 2000, in particular on the period 2005-2010 where this rate reached 46%. This increase in share of Malaysian expansion on peat is also confirmed by the Malaysian Palm Oil Board (Omar et al., 2010) that recorded on the period 2003-2009 a share of 34% of plantation going on peat.[48] Miettinen et al. (2012) found occupation rate even stronger on a short period with 52% (2007-2010), observing very large implantation on peat in Sarawak.[49] In Indonesia, marginal expansion also goes increasingly to peat, with a rate of expansion of 25.4% in 2005-2010, versus 22.2% in 2000-2005. This is mainly driven by a strongly increasing trend in Sumatra (from 28% to 51% in five years) and in Kalimantan (from 5% to 15%). These statistics on the trend of peat disappearance are in line with observation from Miettinen et al. (2012).

Future plantation expansion on peat: in order to project possible rate of expansion into peat, we reviewed four additional studies that project the likely future expansion of oil palm plantations in different provinces. Some works rely on an extrapolation of observed trend on the basis of a detailed spatial analysis (Miettinen et al. (2012)). Some other studies prefer a spatially explicit modelling based on suitability criteria (EPA 2012, Harris et al., 2013). A last approach looks at marginal occupation patterns by assuming areas currently under lease will be developed in the future (Carlson et al., 2012). Policy intervention in the form of a peat moratorium are sometimes considered, as in Harris et al. (2013). However, in the case of this latter scenario, macro-regional land use patterns are not strongly affected.[50] Table 9 provides an overview of the ratio of total future plantation area that is expected to occur on peat according to studies. For Indonesia, the estimations for 2020 range from 13% for the marginal projection of EPA (2012) to 28% for the study of Miettinen et al. (2012). For Malaysia, the lower bound is represented by an estimated 7% for 2020 and 2030 in the BAU scenario by Harris et al. (2013).[51]

Authors usually disagree on the direction of the trend in marginal expansion patterns. EPA (2012) and Harris (2013) keep an assumption of constant rate of expansion of peat, but use a 20 years average and do not take into account the higher levels observed on the decade 2000. Miettinen et al. (2012) and Carlson et al. (2013) assume increasing trend on the basis of recently observed

[47] We rely here on data on planted areas from Gurnaso et al. (2013). It is noteworthy that these statistics based on remote sensing differ from some governmental statistics for planted areas. Areas reported for Malaysia are higher in 2000 (3,467 Mha versus 3,056 Mha for Malaysian department of statistics) and 2010 (5,230 Mha versus 4,202 Mha). For Indonesia, reported statistics are lower for 2000 (3,678 Mha versus 4,158 Mha for Indonesian Ministry of Agriculture) and closer in 2010 (7,724 Mha versus 7,700 Mha).

[48] Using satellite imageries, Omar et al. (2010) report that the average share of plantation on peat increased from 8.2% in 2003 and 13.3% in 2009 in Malaysia, and that 37.4% of plantations in Sarawak were on peat in 2009. This confirms the strong increase on the recent period even if the rate of occupation is slightly lower than in Gunarso et al. (2013) that report 46% for Sarawak in 2010 and 17.8% for total Malaysia.

[49] Miettinen et al. (2012) do not provide statistics on total planted, only on area planted on peat and we use here statistics from Gunarso et al. (2013). In the case of Sarawak, we find that rates in Miettinen et al. (2012) exceeds 100% of marginal expansion on peatland for 2007-2010, which shows some disagreement between the two studies on expansion patterns of plantation in that region.

[50] The moratorium scenario in Harris et al. (2013) affects marginally land use change projections but relies on the assumption that peat conversion will be completely stopped after 2020, both in Indonesia and in Malaysia, while plantations will go on expanding. In our approach, we use for our peatland conversion scenario on less extreme scenarios, relying on historical observations on different periods.

[51] After examinations of spreadsheets from Harris et al. (2013), it was observed that the low rate for Malaysia was due to a cell error and that 7% was used instead of 14% for future expansion.

estimations. Among sources of uncertainty, an important factor is, on the one hand, the localization of future production across provinces with very different patterns, and on the other hand, on the effect of changing policies in each province.

Recalculating marginal expansion rate based on three policy developments: In order to properly disentangle these effects, we apply the scenarios of projections of plantations based on Harris et al. (2013) and Miettinen et al. (2012) across provinces, and assume various development in the trends of marginal expansion in each province. For Harris et al. (2013), we use the business-as-usual scenario, whereas for the approach from Miettinen et al. (2012), we consider two possible linear trends in plantation expansion, that we calculate for the 2000-2010 and the 2005-2010 periods. Last, we also look at what the results would be if the expansion pattern observed over the past years (2012-2013) would continue for the next decade.[52] Projections were considered at the level of the three regions per country, as in Figure 6. The second effect we isolate is the marginal rate of expansion in each region to be applied. We consider three different development in each province:

i) a no regulation scenario ("Trend 10 years"), where the increasing trend on peatland encroachment observed over the past 10 years go on increasing
ii) a stabilization scenario ("Current stable"), where the expansion into peatland remains at the level observed on 2005-2010, without further increase
iii) a policy shift scenario ("Return to hist."), where the trend of expansion into peatland decrease to come back to historical average on the period 1990-2005.

We did not consider scenarios of complete enforced ban of expansion into peatland due the continuation of expansion pressure observed in Indonesia[53] and the high level of opposition to such regulation in the most exposed States, in particular in Sarawak.[54]

[52] We base our analysis of recent statistics on data from the Indonesian official statistics (www.bps.go.id) and the Malaysian Palm Oil Board statistics (http://bepi.mpob.gov.my). According to these statistics, expansion of plantation in Indonesia would have occurred for 60% in Sumatra and for 38% in Kalimantan (in 2013, no data found for 2012). For Malaysia, expansion was 61% in Sarawak and 20% in Malaysian Peninsula and 19% in Sabah. Overall, two third of expansion took place in Indonesia.

[53] USDA reported for the year 2013 10 Mha of oil palm plantation in Indonesia, which challenges optimistic scenarios where production would have declined in most dynamics regions such as Sumatra, an assumption found in Harris et al. (2013) scenarios.

[54] According to the Malaysian Palm Oil Board statistics, plantation expansion in Sarawak would have been 8% in 2013, and total planted area would represent 1.16 Mha in December 2013. Expansion is most likely to continue as the State of Sarawak has announced an objective of 3 million ha. International pressure has been put on Sarawak producers to reduce their expansion into peatland, in particular with the threat of Wilmar, an international oil trader representing half of the Sarawak production purchase to ban palm oil sourced from plantations on peat. The federation of producers (SOPPOA) opposed this measure and still claims 1.2 million ha more peatland with the backing of the government of Sarawak preoccupied by the situation of smallholders. Malaysian producers support a Malaysian standard on palm oil but are critical of standards proposed by the Roundtable of Sustainable Palm Oil, supposed to defend protectionist views of NGOs and to deny possibility of peat agriculture.
Sources: accessed June 2014.
http://www.theborneopost.com/2014/02/15/standing-firm-against-palm-oil-boycott-threat/
http://www.theborneopost.com/2014/01/17/soppoa-wilmars-declaration-detrimental-to-local-industry/
http://www.thestar.com.my/Business/Business-News/2014/02/18/Planters-Its-unfair/
http://www.newsarawaktribune.com/news/22041/SOPPOA-supports-govt-policy-on-oil-palm-devt-in-Sarawak/
http://mypalmoil.blogspot.co.at/2014/04/sarawak-oil-palm-planters-back-mspo.html

The results of our sensitivity analysis on expansion share are presented in Table 10.

As a consequence, we choose to reflect the full range of estimated values in our Monte-Carlo analysis for average expansion and assume the share of plantation going into peat land on the period 2010-2030 to be:

- For Indonesia: **average of 32% (range 11%-57%)**
- For Malaysia: **average of 34% (range 14%-52%)**

These ranges of values both show mean values and uncertainty bounds of comparable magnitude. The uncertainty range covers the historical rates observed on the period 2005-2010 (25.4% for Indonesia and 46% for Malaysia, due to the strong surge in the Sarawak state[55]).

3. Implication for model results

Alongside emission factors for drained peat, these estimations of the expansion patterns into peat will be used to calculate in the model a plausible range of total emissions attributable to oil palm production in Indonesia and Malaysia. No peat land emissions will be considered for other regions than Southeast Asia due their more marginal contribution to overall wetlands emissions associated to palm oil.

Possibility of other outlets than Western world
http://www.thestar.com.my/Business/Business-News/2014/03/10/Sarawak-plans-to-sell-CPO-in-Middle-East/

[55] If trends observed on the period 2005-2010 were to continue in Sarawak, palm plantation could convert the total initial peat area in that State, i.e. 1.3 to 1.4 Mha (according to Gurnaso et al. (2013) and Mittienen et al. (2012), respectively), by the end of the decade 2020. Factoring in this consideration in the calculation leads to a lower rate for the subsequent period (2020-2030), which explains that the average rate over 2010-2030 hardly exceeds 50% in our estimation.

Table 8 : Estimates of total oil palm planting area in 2010 (in ha), oil palm planting area on peat (in ha) and % of total planting area by region for Indonesia and Malaysia according to four studies with varying coverage.

Source	Gunarso *et al.* (2013)			Koh et al. (2011)			Carlson et al. (2012)			Miettinen *et al.* (2012)	
Region	Total planting area (ha)	Planting area on peat (ha)	Share on peat	Total planting area (ha)	Planting area on peat (ha)	Share on peat	Total planting area (ha)	Planting area on peat (ha)	Share on peat	Planting area on peat (ha)	Share on peat*
Sumatra	4,743,308	1,395,733	29%	3,871,839	464,554	12%	n.d.	n.d.		1,047,000.00	22%
Kalimantan	2,896,952	307,515	11%	1,100,105	43,184	4%	3,164,005	402,166	13%	314,000.00	11%
Papua	83,622	1,727	2%	n.d.	n.d.		n.d.	n.d.		n.d.	
Total Indonesia	**7,723,882**	**1,704,975**	**22%**	**4,971,944**	**507,738**	**10%**	**n.d.**	n.d.		**1,361,000**	18%
	Gunarso *et al.* (2013)			Koh et al. (2011)			Omar et al. (2010) (data from 2008/09)			Miettinen *et al.* (2012)	
Peninsular Malaysia	1,510,809	215,984	14%	2,005,833	236,820	12%	2,503,682	207,458	8%	238,000	16%
Sarawak	1,033,260	475,946	46%	357,915	103,841	29%	1,167,172	437,174	37%	494,000	48%
Sabah	1,510,809	29,028	2%	918,739	30,166	3%	1,340,317	21,405	2%	50,000	3%
Total Malaysia	**4,054,878**	**720,958**	**18%**	**3,282,487**	**370,827**	**11%**	**5,011,171**	**666,038**	**13%**	**782,000**	**19%**

***** Calculated from Gurnaso et al. (2013) data on total oil palm plantation areas.

Source: authors' compilation.

Table 9 : Overview of studies that project oil palm expansion on peat in the future.

Study (year)	Affiliation / funder	Methodology	Study area	Percent of plantations on peat (historical)		Percent of plantations on peat In 2020 (2030)		Underlying assumptions	Comment
				Indonesia	Malaysia	Indonesia	Malaysia		
Harris et al. (2013)	Roundtable for Sustainable Palm Oil	GEOMOD	Indonesia, Malaysia, Papua New Guinea	22%	18%	22% (22%)	7% (7%)	Historical average (constant rate of plantations on peat)	
				22%	18%	19% (17%)	13% (12%)	Peat Moratorium (no further expansion)	
Miettinen et al. (2012a, 2012b)	Univ. Singapore / International Council on Clean Transportation	Extrapolation from spatial trends	Indonesia, Malaysia	18%	19%	28%	42%	Linear projection based on 2007-2010 period	
EPA (2012)	U.S. Environmental Protection Agency	GEOMOD	Indonesia, Malaysia	22%	13%	15%	10%	Historical projection	Sensitive to the ratio of mature – immature palms Projected to 2022
		GEOMOD	Indonesia, Malaysia			13%	9%	Projected incremental expansion	
Carlson et al. (2012)	Univ. Yale & Stanford	Static model	Kalimantan	13%	n.d.	17%	n.d.	Development of all oil palm leases issued until 2012 by 2020	

Source: authors' compilation.

Table 10 : Projected expansion on peatland according to literature land use scenarios and local patterns of expansion on peat.

	Indonesia 2010-2030			Malaysia 2010-2030		
	Local expansion pattern on peat			Local expansion pattern on peat		
Regional land use scenario	Trend 10 yrs	Current stable	Return to Hist	Trend 10 yrs	Current stable	Return to Hist
Proj. Harris et al. (2013)	31%	17%	6%	45%	44%	24%
Proj. Linear 2000-2010	51%	31%	14%	36%	32%	14%
Proj. Linear 2005-2010	44%	25%	11%	41%	38%	18%
Proj. Linear 2012-2013	57%	38%	19%	52% [b]	49%	20%
Min	**11%** [a]			**14%**		
Mean	**32%** [a]			**34%**		
Max	**57%** [a]			**52%**		

[a] Projections from Harris et al. (2013) were leading to lower expansion into peatland due to a slowing down of production in Sumatra Island. However, the recent trends show that such projections were not realistic, as expansion of palm oil has reached 6.6 Mha in 2013 according Indonesian official statistics. For that reason, we did not keep results based on Harris et al. (2013) projections for our summary statistics of Indonesia.

[b] This scenario leads to complete use of peatland in Sarawak by 2030, which decrease the expansion rate into peatland in that region at the end of the period. See note 55.

Source: authors calculation.

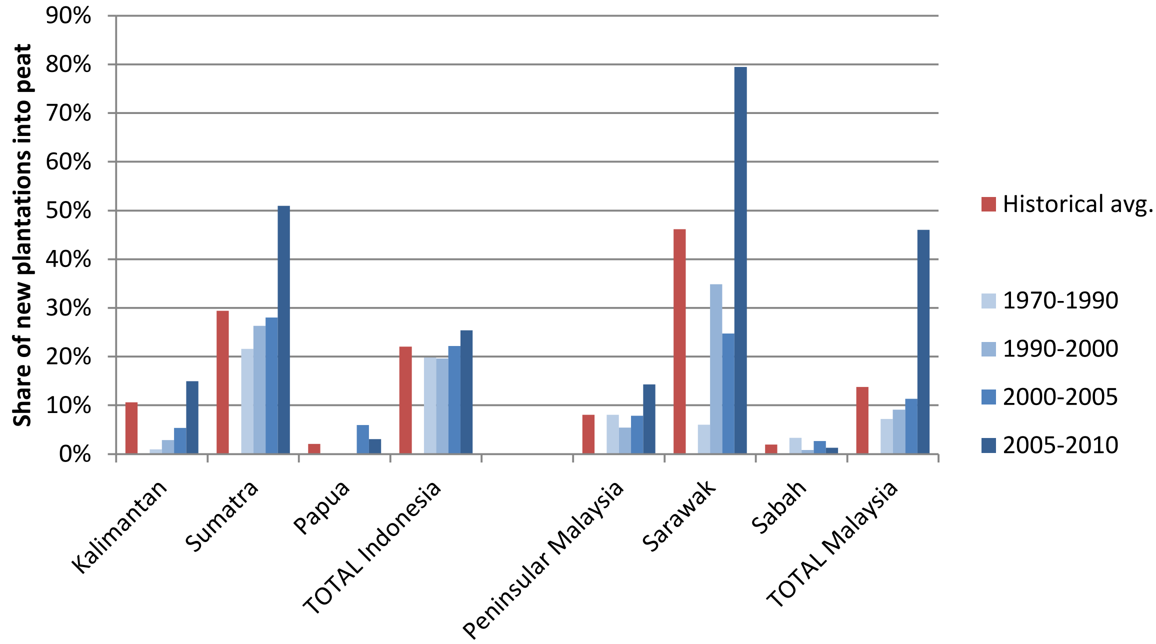

Figure 6 Percentage expansion of plantation into peat land in different regions of Indonesia and Malaysia over time. Source: Gurnaso et al. (2013).

Improvement 8: Expand inclusion of soil organic carbon to rest of the world

1. Motivation for improvements

Soil organic carbon (SOC) is a prominent C stock and its representation is important for a comprehensive accounting of GHG emissions from agriculture and land use change. In previous versions of GLOBIOM, a refined SOC accounting design had been developed for the European Union. As indirect land use change from biofuels occurs in- and outside of the EU, there was a need to expand the representation of SOC accounting in the model to other regions of the world.

2. Methodological approach

We complemented the initial dataset of soil organic carbon in GLOBIOM with data from the Harmonized World Soil Database v1.2 (HWSD, see FAO et al., 2012). This database is a spatially explicit layer of soil information in the different regions of the world, such as organic carbon, water storage capacity, soil depth etc. The information on SOC is here used as input in GLOBIOM at the grid level. This database therefore complements the EU datasets already in the model (Lugato et al., 2013 for cropland, Jones et al., 2005 for other land use types). A summary of average SOC content by large region and land use type is displayed in Table 11.

In order to track in the model changes in SOC content in the different regions, we then applied a Tier 1 approach[56] based on GHG accounting IPCC guidelines (IPCC, 2006). The formula applied is as follows (Equation 2.25):

$$SOC = \sum_{(c,s,i)} (SOC_{REF} * F_{LU} * F_{MG} * F_{I} * A) \text{, where:}$$

c, s, i are respectively the climate zones, soil types and management systems in the region

SOC_{REF} is the carbon stock of reference, calculated using the initial 2000 data and the initial management information

F_{LU} is the land use factor informing on type of use among crop cover, flooded areas for rice, perennial crops, or set aside land;

F_{MG} is the management factor informing on tillage practice

F_{I} is the input factor informing on level of fertilizer input and use of manure

A is the land use area for climate zone c, soil type s and management type i.

Default values of F_{LU}, F_{MG}, F_{I} are sourced from Table 5.5 and Table 6.2 of IPCC guidelines. SOC_{REF} is determined in our setting on the basis of the HWSD and EU specific datasets, correcting for the land use, management and input factors when relevant. Therefore, we do not use here the averaged IPCC default value, in order to fully benefit from the spatially explicit information. Differences between IPCC reference SOC values and HWSD values are documented in Carré et al. (2010).

[56] Tier 1 approach corresponds to the default methodology proposed by IPCC guidelines, when more local information is not available.

Land use change and management information (crop type, fertilizer level) are calculated endogenously in the model. Tillage practice is another important component determining level of SOC. Because tillage practice is not explicitly modelled at the global level in GLOBIOM, we assume for most regions a full tillage practice for cropland, except for the EU and some large countries where more precise management information could be retrieved (USA, Brazil, Argentina). The tillage assumptions are summarized in Table 12. One additional management changed is considered for the agricultural residues where the effect of removing residues on C stock is accounted for (see Improvement 1).

In order to keep the full consistency between SOC in Europe and Rest of the World, and prevent asymmetrical calculations, the same IPCC Tier 1 formula is here applied for EU and for the Rest of the World.

3. Implication for model results

Additional SOC at world level in the accounting will allow a more comprehensive coverage of indirect emissions related to expansion of agricultural land. The approach remains here however simplified, based on a Tier 1 approach but follows recommendation from JRC on accounting of soil organic carbon (Carré et al., 2010). A limitation of the approach is the impossibility to consider C stock change associated to restoration of degraded land, due to the too limited data on C stock and locations of degraded areas at world level.

Table 11. Reference level for soil organic carbon in t C/ha by GLOBIOM region. Reference levels correspond to C stocks in an undisturbed grassland area.

	SOC_{REF}
Australia / New Zealand	20
Argentina	27
Brazil	34
Canada	58
China	33
Congo Basin	35
Former USSR	48
India	27
Indonesia	48
Japan	56
Malaysia	34
Mexico	29
Mid-East North Africa	17
Pacific Islands	39
Rest of Central America	40
Rest of Eastern Europe	34
Rest of Western Europe	58
Rest of South America	35
Rest of South Asia	22
Rest of Southeast Asia	32
South Africa	20
South Korea	33
Eastern Africa	26
Southern Africa	24
Western Africa	22
Turkey	26
Ukraine	47
USA	35
EU28	59

Note: Reference SOC level is calculated based on HWSD, except for the EU where a more detailed map is used (Jones et al., 2005). In that latter case, it has been calculated as the average SOC stocks on forests, pastures and other natural vegetation.

Table 12. Average tillage assumption for GLOBIOM regions in the base year

Region	Share Full tillage	Share reduced tillage	Share no tillage
European Union	76%	18%	6%
Canada	40%	30%	30%
USA	75%	--	25%
Brazil	65%	--	35%
Argentina	30%	--	70%
Paraguay	28%	--	72%
Australia	75%	--	25%
Rest of the world	100%	--	--

Source: PICCMAT project data for the European Union http://climatechangeintelligence.baastel.be/piccmat/ , Canadian Agricultural Census for Canada, Derpsch et al. (2010) for other regions.

Improvement 9: Representing forest regrowth and carbon reversion

1. Motivation for improvement

GLOBIOM usually assumed that, when cropland or grassland is abandoned, it can either be turned into managed forest or forest plantation, if economically profitable, or abandoned into other natural vegetation. Carbon stock in living biomass for other natural vegetation was taken from the Ruesch and Gibbs (2008) database. However, even if not actively managed as forest or plantation, abandoned land can lead to progressive forest regrowth in some regions, with carbon stock higher than typically assumed for natural grassland, for instance; it was therefore decided to implement a more detailed representation. The question of the timing of this carbon sequestration is also important, because the model solves by 10 year time steps and the evaluation period of the policy is limited to 20 years.

2. Methodological approach

IPCC (2006) guidelines distinguish two types of carbon reversion associated to forest regrowth, natural and artificial regeneration.[57] Considering that managed forests are already represented in GLOBIOM, we only look here at the carbon accumulation in land converted into unmanaged forest. The typical type of conversion of land to Forest is assumed according to IPCC to have an average transition period of 20 years, after which the land can be considered classified as Forest again. IPCC default table provides a single value per type of forest for different regions (see overview in Table 13). However, GLOBIOM contains geographically information on the carbon stocks in natural forest for each Production Unit in the model. We therefore use the geographical heterogeneity of stock distribution to vary geographically the default value for carbon growth.

However, forest regrowth does not systematically take place when land is abandoned, because of various environmental factors. Indeed, the abandoned land can be too infertile or degraded, as indicated by IPCC guidelines.[58] A land classified as “Other land” (i.e. not cropland, grassland, forest land, wetland or settlement) is under IPCC accounting rule considered by default not accumulating carbon (see IPCC, 2006, Vol.4, Chap. 9). To distinguish what share of land can be considered returning to forest or only to other natural vegetation (with constant carbon stock), we apply the same method as suggested by the U.S. Environmental Protection Agency (EPA, 2010). Within each Production Unit, we

[57] “Land is converted to Forest Land by afforestation and reforestation, either by natural or artificial regeneration (including plantations). The anthropogenic conversion includes promotion of natural re-growth (e.g., by improving the water balance of soil by drainage), establishment of plantations on non-forest lands or previously unmanaged Forest land, lands of settlements and industrial sites, abandonment of croplands, pastures or other managed lands, which re-grow to forest. […] Land conversion may result in an initial loss of carbon due to changes in biomass, dead organic matter, and soil carbon. But natural regeneration or plantation practices lead to carbon accumulation and that is related to changes in the area of plantations and their biomass stocks.” [IPCC (2006) Guidelines, Vol 4., Chapter 4, p. 4.30]

[58] “Some abandoned lands may be too infertile, saline, or eroded for forest re-growth to occur. In this case, either the land remains in its current state or it may further degrade and lose organic matter. Those lands that remain constant with respect to carbon flux can be ignored. However, in some countries, the degradation of abandoned lands may be a significant problem and could be an important source of CO2. Where lands continue to degrade, both above-ground biomass and soil carbon may decline rapidly, e.g., due to erosion. The carbon in eroded soil could be re-deposited in rivers, lakes or other lands downstream. For countries with significant areas of such lands, this issue should be considered in a more refined calculation.” [IPCC (2006) Guidelines, Vol 4., Chapter 4, p. 4.30]

assume that, forest regrowth takes place at the same share on abandoned land as is the share of forest already observed on fertile land (forest and other natural vegetation).

3. Implication for model results

Carbon balance in the model is changed with this improvement, with better representation of carbon sequestration when land is abandoned. Former and new carbon stocks associated to abandoned land are summarized in Table 14. In the former accounting approach, other natural land carbon stock was allocated to abandoned land using data from Ruesch and Gibbs but without any dynamic consideration. Therefore, after 10 years, the carbon stock was already maximum, a pattern that is changed with the new approach, which decreases the carbon stock typically reached after 10 years. Additionally, the new approach considers possibility to reach higher level of stock directly through forest regrowth (e.g. in Europe).

Table 13. Carbon accumulation in living biomass for natural forest regrowth in different regions (first 20 years)

Region	Ecological zone	Average above-ground biomass growth (tonnes d.m. / ha / yr)	Ratio below-ground to above ground biomass [a]	Average above and below ground C accumulation (tonnes C / ha / yr)
Tropical and subtropical zones				
Africa	Tropical rainforest	10	0.37	6.4
Africa	Tropical moist deciduous forest	5	0.20 – 0.24	7.5
Africa	Tropical and subtropical dry forest	2.4	0.28 – 0.56	1.6
Africa	Tropical shrubland and subtropical steppe	0.2 – 0.7	0.32 – 0.4	0.3
Africa	Tropical and subtropical montain systems	2.0 – 5.0	0.27	2.1
Asia (continental)	Tropical rain forest	7.0	0.37	4.5
Asia (continental)	Tropical moist deciduous and subtropical humid forest	9.0	0.20 – 0.24	13.5
Asia (continental)	Tropical and subtropical dry forest	6.0	0.28 – 0.56	4.0
Asia (continental)	Tropical shrubland and subtropical steppe	5.0	0.32 – 0.4	3.2
Asia (continental)	Tropical and subtropical mountain systems	1.0 – 5.0	0.27	2.1
Asia (insular)	Tropical rain forest	13	0.37	8.4
Asia (insular)	Tropical moist deciduous and subtropical humid forest	11	0.20 – 0.24	16.5
Asia (insular)	Tropical and subtropical dry forest	7.0	0.28 – 0.56	4.7
Asia (insular)	Tropical shrubland and subtropical steppe	2.0	0.32 – 0.4	1.3
Asia (insular)	Tropical and subtropical mountain systems	3.0 – 12	0.27	4.5
North America	Tropical rain forest	0.9 – 18	0.37	6.1
South America	Tropical rain forest	11	0.37	7.1
North and South America	Tropical moist deciduous and subtropical humid forest	7.0	0.20 – 0.24	10.5
North and South America	Tropical and subtropical dry forest	4.0	0.28 – 0.56	2.7
North and South America	Tropical shrubland and subtropical steppe	4.0	0.32 – 0.4	2.6
North and South America	Tropical and subtropical mountain systems	1.8 – 5.0	0.27	2.0
Temperate zones				
Europe	Temperate oceanic forest	2.3	0.40 – 0.46	1.5
Europe	Temperate continental forest	4.0	0.40 – 0.46	2.7
Europe	Temperate mountain systems	3.0	0.40 – 0.46	2.0
North America	Temperate oceanic forest	15	0.40 – 0.46	10.1
North America	Temperate continental forest	4.9	0.40 – 0.46	3.3
North America	Temperate mountain systems	3.0	0.40 – 0.46	2.0
South America	Temperate oceanic forest	2.4 – 8.9	0.40 – 0.46	3.8
Asia	Temperate continental forest	4.9	0.40 – 0.46	3.3
Asia	Temperate mountain systems	3.0	0.40 – 0.46	2.0
Boreal zones				
Asia, Europe, North America	Boreal coniferous forest	0.1 – 2.1	0.39	0.7
Asia, Europe, North America	Boreal tundra woodland	0.4	0.39	0.3
Asia, Europe, North America	Boreal mountain systems	1.0 – 1.1	0.39	0.7

[a] For the below to above ground ratio, the coefficient selected correspond to biomass density below or equal to a growth period of 20 years.

Source: IPCC (2006), Vol. 4, Chap. 4, Table 4.4 and Table 4.9. Last column calculated using carbon fraction value of 0.47.

Table 14. Average carbon stock from sequestration in abandoned land in old and new GLOBIOM approach (tonnes C/ha). Weighting within each region is done by agricultural area, as this is the areas of particular interest for abandonment.

Region	Carbon stock sequestered after abandonment (old approach, constant)	Carbon stock after 10 years (new approach)	Carbon stock after 20 years (new approach)
World	47	20	40
Middle East North Africa	24	11	21
Sub-Saharan Africa	63	25	51
Former Soviet Union	9	10	19
Latin America	55	24	47
North America	29	16	33
South Asia	52	19	39
Europe	29	26	53
Oceania	49	20	40
Eastern Asia	37	9	18
Southeast Asia	59	24	48

Improvement 11: Representation of biofuel co-product in GLOBIOM

1. Motivation for improvements

Co-products are an essential component of the lifecycle analysis of biofuel production. Biodiesel from rapeseed, soybeans, and sunflower leads to significant amounts of protein meals being delivered on the market. These products are used in animal diet as protein supplement and the biofuel sector strongly interacts with the livestock sector through this channel. Similarly, dried distillers' grains with solubles (DDGS) and sugar beet fibers, generated by cereals and sugar beet processing respectively, are also used for feed preparation and displace consumption of other agricultural products. To understand the final balance associated to these changes, it is important to understand which products are being substituted to which extent through increased supply of co-products from the biofuel processing chain. The present improvement focuses on the representation of the feed nutrients in terms of energy and proteins, and complements efforts on the modeling of oilseeds markets from improvement 21 on substitution mechanisms and improvement 34) and 35) on oilseeds transformation chains.

2. Methodological approach

GLOBIOM already incorporates a precise description of animal diets for each livestock system and species, which can be used to best represent the substitution patterns for biofuel coproducts. We calculate for each species modified diet specifications incorporating more co-products, subject to some maximum incorporation constraints.

Diet specifications are calculated taking into account digestibility patterns and metabolisable energy and proteins for the different animal types. Feed requirements are calibrated on the current data used in GLOBIOM, and derived from the RUMINANT model (Herrero et al., 2013). For each feed item, we calculate the exact nutrient content using feed tables from U.S. National Research Council (NRC, 1982), as presented in Table 15.[59] These tables contain all major crop types traditionally used for feeding, including protein meals and DDGS. In the case of DDGS, however, as technology evolved a lot over time in terms of protein extraction efficiency, we used more recent data on DDGS characteristics. Because composition of coproducts can vary across places and refineries, sensitivity analysis will also explore slight variation around the values presented in Table 15.

This information on feed composition is used to specify substitution patterns for each animal type, by ensuring that both the energy and protein balance are preserved. This diet substitution pattern is applied only to the livestock systems based on grain and protein meals consumption (of type intensive, mixed-intensive or mixed-extensive). Substitution with co-products for grass-based production systems is not considered. To represent substitution, we allow mobility in the feed intake of the animals while satisfying two inequalities directly coded in the model: 1) the crude protein intake should be higher or equal to initial intake; 2) the metabolisable energy intake should be higher or equal to initial intake. These equations are applied to protein meals and to the main feed grain type used in the region (usually

[59] Although this source might appear old, changes in feedstuff nutrient composition remained limited, as illustrated by a comparison with more recent tables (for instance for beef, NRC, 2000).

corn or wheat), to adjust on energy content. For cattle, where we have more detailed information, we directly use the average of Net energy for growth and for maintenance, whereas for dairy cows, we refer to the Net Energy for lactation to capture the specific dietary needs in the respective livestock sectors.[60]

Because each species and each feedstuff have different characteristics, the replacement results used differ depending on the livestock sector and the biofuel pathway. The advantage of this approach is to be able to directly trace the substitution efficiency on the basis of nutrient content of co-products, instead of relying on substitution coefficients from the literature.

Table 16 shows the substitution patterns obtained when applying the composition found in Table 15. We calculate in this table a simple bilateral substitution between one biofuel by-product and two feed products. Sign indicate if the co-product replaces a feedstuff (positive) or requires an additional provision of cereals to preserve the energy balance (negative). For example, one unit of rapeseed meal for beef triggers an additional consumption of 0.090 unit of corn while substituting 0.833 unit of soya meal. Because several protein meals can substitute with each other, some more complex substitution can also appear. For instance, wheat DDGS can substitute with soybean meal - and with some cereals to satisfy the complete energy balance - but soybean meal can also in turn substitute with some other oilseed meals. Table 16 is therefore only illustrative of the simplest substitution patterns with a pair of feed products.

In complement to substitution possibilities, for each animal type, incorporation of DDGS is limited due to the nutrient characteristics of co-products, some of them not directly accounted for in the model substitution patterns. In particular, DDGS too high incorporation rates can lead to an oversupply of proteins and phosphorus, leading to waste disposal issues that affect manure management (Hoffman & Baker, 2011). For DDGS, we therefore capped the incorporation levels at some selected values on the basis of a literature review by Hoffman and Baker (2011). These incorporation constraints are provided in

[60] Different beef and dairy cattle have different feed requirements. Maintenance energy intake is required for all animals to ensure the appropriate level of feed for normal metabolism, at equilibrium, without production of any other output. For beef cattle, it needs to be supplemented by energy for growing with different ratios depending on the stage of development of the animal. For dairy cows, milk production requires an additional regular intake of energy that leads to the lactation energy requirement (that includes maintenance energy).

Table 17 and were chosen at the mid-range of low and high values in the literature, except when higher incorporation rates were already observed in U.S. statistics (beef cattle).

3. Implication for model results

The new representation of co-product substitution in the model will allow to specify the substitution of animal feeding with different diet possibilities specific to each species, on the direct basis of nutrient composition and their properties per type of animal. For instance, wheat DDGS will substitute more cereals in the ruminant sector than with the non-ruminant, and sugar beet pulp will be little used by the poultry sector due to poor digestibility. We also observe that, as long as other nutrient constraints (e.g. amino-acids) are not taken into account, rapeseed and sunflower meal can appear as appealing substitutes with other protein sources due to their high protein level, but may require cereals complement to preserve the energy balance, which was not captured before. This new design will allow for a more accurate accounting of the substitution possibilities of co-products in the feed and to better measure the land use change effects implications associated their incorporation.

Table 15. List of metabolisable energy and protein content associated to the different feed crops and supplements in the model. All values below are expressed for dry matter feed. ME = metabolisable energy, NEm = net energy for maintenance, NEg = net energy for growth, NEl = net energy forlactation, MEn = metabilisable energy, nitrogen corrected (for poultry).

Feed stuff	Ruminant ME (Mcal/kg)	Ruminant NEm(Mcal/kg)	Ruminant NEg(Mcal/kg)	Dairy cattle NEl(Mcal/kg)	Chicken MEn (kcal/kg)	Swine ME (kcal/kg)	Crude protein (%)	Crude Fiber (%)
Barley grain	3.29	2	1.35	1.94	2843	3299	13.5	5.7
Dry bean	3.29	2	1.35	1.94	2593	3.772	25.3	5
Corn grain	3.42	2.09	1.42	2.01	3818	3724	10.9	2.9
Corn silage	2.62	1.55	0.94	1.57	NA	2981	8.3	25.1
Oats grain	2.98	1.79	1.17	1.77	2862	3012	13.3	12.1
Pea	3.42	2.09	1.42	2.01	2385	3416	25.3	6.9
Potato	3.16	1.91	1.27	1.87	NA	3516	9.5	2.4
Rapeseed meal solv extd	2.62	1.55	0.94	1.57	1924	2935	40.6	13.2
Rye, grain	3.29	2	1.35	1.94	3001	3327	13.8	2.5
Soybean seeds	3.60	2.22	1.52	2.11	3674	3905	42.8	0.1
Soybean meal solv extd, 44% protein	3.29	2	1.35	1.94	2485	3155	49.9	7
Soybean oil	8.23	5.25	4.02	4.66	8667	7283	1.4	NA
Sugar beet pulp, with molasses, dehydrated	2.93	1.76	1.14	1.74	719	3139	10.1	16.5
Sunflower meal, wo hulls, meal solv extd	2.45	1.44	0.82	1.47	2242	2851	49.8	12.2
Triticale grain	3.29	2	1.35	1.94	3521	3396	17.6	4.4
Wheat grain	3.47	2.12	1.45	2.04	3401	3660	16	2.9
Wheat durum grain	3.34	2.03	1.37	1.96	3652	3492	15.9	2.5
Source : National Research Council, 1982, 2000								
Corn distillers grains with solubles, dehydrated	**3.90**	**2.38**	**1.69**	**2.28**	**2531**	**3790**	**31.2**	**8.6**
Wheat distillers grains with soluble, dehydrated	**3.75**	**2.29**	**1.62**	**2.14**	**2406**	**3472**	**36.6**	**7.6**
Sources : Wheat DDGS : Noblet, Cozannet & Skiba (2012) for pigs and poultry; extrapolated from Kalscheur et al. (2012) for ruminant. Corn DDGS: Kalscheur et al. (2012) for ruminant; extrapolated from Anderson et al. (2012) for pigs and Noblet, Cozannet & Skiba (2012) for poultry.								

Table 16. Substitution pattern for each animal species for one unit of coproduct consumed by the livestock sector. Positive values correspond to a replacement of feed, negative value to a joint addition of another feedstuff to preserve the energy balance.

	Corn DDGS	Wheat DDGS	Sugar beet pulp	Rapeseed meal	Sunflower meal
Feed item					
SUBSTITUTE FOR CORN & SOYBEAN MEALS					
Beef					
Corn	0.711	0.523	0.800	-0.085	-0.390
Soya meal*	0.470	0.619	0.028	0.832	1.083
Dairy					
Corn	0.673	0.452	0.849	-0.005	-0.294
Soya meal*	0.478	0.635	0.017	0.815	1.062
Swine					
Corn	0.559	0.382	0.824	0.121	-0.098
Soya meal*	0.494	0.650	0.022	0.787	1.019
Poultry					
Corn	0.298	0.178	0.066	-0.030	-0.073
Soya meal*	0.560	0.695	0.188	0.820	1.014
SUBSTITUTE FOR WHEAT & SOYBEAN MEALS					
Beef					
Wheat	0.791	0.582	0.890	-0.094	-0.434
Soya meal*	0.371	0.547	-0.083	0.844	1.137
Dairy					
Wheat	0.753	0.506	0.950	-0.006	-0.329
Soya meal*	0.384	0.571	-0.102	0.816	1.103
Swine					
Wheat	0.686	0.437	0.944	0.139	-0.112
Soya meal*	0.405	0.593	-0.100	0.769	1.034
Poultry					
Wheat	0.375	0.224	0.083	-0.038	-0.091
Soya meal*	0.505	0.662	0.176	0.826	1.027

Note: Soybean meals are not represented here as they are largely used already as feed in rows. Their substitution values can be found by reading the table from row to column and inverting the cereal contribution. For instance, 0.833 unit of soybean meal substitute for beef with 1 unit of rapeseed meal and 0.09 unit of corn (corn is now replaced as the negative sign needs to be inversed).

Table 17. Maximum incorporation constraint for DDGS as percent of daily dry matter intake.

Animal type	Observed incorporation rate	Maximum incorporation in literature		Value in GLOBIOM
	US Widwest (2007)	Low	High	
Beef [a]	22%	10%	30%	30% [b]
Dairy [c]	8%	10%	30%	20%
Swine	10%	10%	30%-50% [d]	20%
Poultry	NA	10%	15%	12.5%

[a] Statistics reported here are based on calculation for cows. Beef cattle on feed high bound up to 40%.

[b] High bound taken to take into account observed rate.

[c] Incorporation statistics reported here for dairy cows (not replacement heifers).

[d] Low bound for market swine, high bound for breeding swine.

Source: Hoffman and Baker (2011).

Improvement 15: Representing multi-cropping in GLOBIOM

1. Motivation for improvement

In several regions of the world, the possibility of harvesting more than one crop per year in a same field has been used to increase output per hectare. Most famous examples are the multi-harvest of rice in Southeast Asia or the soybean-maize double cropping practice in Latin America. GLOBIOM was not taking into account so far this possibility and annually harvested areas of cropland were calculated on the basis of harvested areas of each crop, without any specific correction. Multi-cropping (or inter-cropping) possibilities were therefore not considered. Additionally, when cropland area was found larger than harvested areas, the "unused" cropland was considered kept constant, to reflect the presence of other not referenced crops or various conservation uses. No change in cropland harvest frequency was then represented. Therefore, it was decided to better represent the trend of multi-cropping in the baseline of GLOBIOM by introducing some cropping intensity change and the potential of this development to free some agricultural land.

Useful definitions [We largely base our definitions here on Ray and Foley (2013)]

Harvested area: Area of crop that has been harvested through one year, possibly several time in case of successive cropping seasons in the same year.

Annually harvested cropland: Area of cropland which is used for cultivation (possibly several times a year).

Total standing cropland: total area of land declared as cropland, including fallow land.

Cropland harvest frequency (CHF) or cropping intensity (depending on author): defined as *Harvested area* divided by *Total standing cropland*.

Double-/Multi-cropping: practice of harvesting two/several crops successively in a same year on the same cropland area.

Inter-cropping: practice of planting several crops simultaneously in the same field, with alternate rows of crop of the different species.

2. Methodological approach

First, FAO statistics were used to calculate cropland harvest frequency (CHF) of the different regions. CHF calculation is not sufficient to identify all places where muti-cropping could be observed due to disparity of cropping practices within a region. However, it provides a sufficient criteria to locate some of them. Indeed, if this ratio is greater than one, some areas of land have necessarily been used to grow several crops in the same year. If this ratio is below one, a share of cropland has necessarily not been used for production, but this does not exclude multi-cropping practices in some other locations in the region.

Nine countries were found with CHF > 1 for the calibration year (2000), which reveals the presence of multi-cropping in these regions. The list of regions and corresponding countries can be found in Table 18. They are consistent with assessment of multi-cropping location in the literature (Langeveld et al., 2013; Siebert et al., 2010). For these regions, the yield assumptions for rice were adjusted in GLOBIOM to better reflect the current average output per hectare of harvested cropland and per year. As a consequence, cultivated areas were decreased in these regions, and land areas erroneously allocated to the cultivation of the crop were reclassified as “other natural vegetation”.

For regions with CHF >1, we also implemented a trend in the baseline for cropland harvest frequency. For this we used the trend on the period 2000-2011, following a methodology similar to Ray & Foley (2013), who have calculated trends in ratio of harvested land over cropland area over time.[61]

For regions with CHF<1, it is not possible to derive frequency of harvest for the different crops, without studying some specific national datasets, a process too time consuming for this project. As an exception to this general rule, we had a closer look on the case of Brazil, well known for his increasingly use of multi-cropping practices (on corn and soybeans). According to Spera et al. (2014), use of double cropping in the State of Matto Grosso grew from 500,000 ha in 2001 to 2.8 Mha in 2011. Therefore we also apply the trend on cropland harvest frequency in Brazil (+0.9% per year on frequency), although the cropland harvest frequency is lower than 1 in this region.

3. Implication for model result

The first effect of multi-cropping representation will be, for the countries with CHF > 1, a reevaluation of annually harvested cropland area for the base year 2000, after adjusting the yield values. Additionally, the trend on yield will be modified for these regions as well as for Brazil. So far, the exogenous yield trend was only representing the effect of technical change, but it will now also incorporate an additional component for change in management related to multi-cropping. Yields are likely to grow faster for the regions concerned, reducing the impact of additional production in the baseline on land use change.

[61] It should be noted that the trend in historical data may also be associated to change in fallow land (decreasing) and not only to multi-cropping. However, as unused cropland is kept constant in the model (see improvement 27 for more explanations), this approach is consistent to replicate the trend in cropland harvest frequency.

Table 18. Cropland harvest frequency and associated indicators according to FAOSTAT in 1999-2001.

Region with multi-cropping	Harvested area – cropland (1000 ha, only >1Mha reported)	Cropland harvest frequency (2000)	Annual growth rate (2000-2011)	Maximum cropland harvest frequency (Ray and Foley, 2013)
China	29,089	1.22	1.4%	1.02
Nigeria	8,537	1.26	-1.7%	2.00
India	6,514	1.04	1.2%	1.63
Bangladesh	5,544	1.63	1.1%	1.99
VietNam	3,865	1.47	-0.5%	1.95
Philippines	2,779	1.28	0.2%	2.00
Myanmar	2,551	1.24	1.6%	1.80
Nepal	2,052	1.84	0.8%	1.06
Egypt	1,271	1.38	0.5%	1.01
Others (<1 Mha)	1,347	1.02	--	--
TOTAL	**63,549**	**1.13**	**--**	**--**
Brazil		***0.78***	0.9%	1.71
World		***0.82***		

Source: authors' own calculation using FAOSTAT data, except for last column from Ray and Foley (2013). Note that we report here growth rates for the index, whereas Ray and Foley report rate of annual change.

Improvement 21: Representing vegetable oil substitution in GLOBIOM

1. Motivation for improvements

Vegetable oils were represented in the standard version of GLOBIOM with distinct demand functions, and the level consumed was only determined by an exogenous shifter for the income effect and an own-price demand elasticity for the price effect. No distinction was made between demand for food and demand for industrial use, to the exception of biofuel use. However, vegetable markets are to some extent connected, as illustrated by the strong correlation between the different oil prices. It was agreed that a better representation of these linkages should be introduced into GLOBIOM, by introducing some substitution possibilities between vegetable oil on the supply side, while keeping in mind the restrictions to such substitution related to the different properties of these oils, the specific needs of industries, as well as the preferences of consumers.

2. Methodological approach

The question of patterns of change in the oilseed market is complex and we investigated different sets of statistics to better understand the mechanisms at play, provided by the industry, by FAOSTAT and by the USDA.[62] Stylized facts were examined to address a certain number of questions in the debate. We observed the following points:

- Food consumption per capita of vegetable oil has been relatively stable in Europe for rapeseed over the past decade (-9% according to USDA). But sunflower oil consumption as food has notably increased between 2002 and 2012 (USDA: +38%), as well as palm oil (+63% between 2000 and 2012). Soybean oil has decreased in the same time by 30% according to USDA (See Figure 7). In the EU, the use of soybeans and rapeseed has decreased to the benefit of sunflower and palm oil.
- Most of substitution in the EU on vegetable oils has been observed through imports and on the industrial uses market. By contrast however, the substitution patterns in the US were larger for final consumption, and soybean oil used as food was significantly substituted by palm oil (Figure 8).
- Decrease in EU food consumption of rapeseed has remained limited compared to total increase in supply (see Figure 7 and Figure 8). The main sources of additional supply have been increased rapeseed production, and additional rapeseed imports (see Figure 10).
- Palm oil imports to the EU have significantly expanded over the period 2000-2012 (see Figure 10). Parts of these imports have been driven by a direct use by the industrial sector, in particular biofuels. But one third of these imports have also been absorbed by the food sector. The food

[62] Methodology applied by USDA to split consumption across uses was found more consistent with the data provided by the industry than the one from FAOSTAT. The latter allocate a large part of consumption to other uses, whereas such use is usually better allocated in USDA and FEDIOL databook. Unfortunately, the USDA data do not provide the decomposition of demand between the different industrial uses. Therefore, the substitutability of vegetable oil within the non-biofuel industrial uses is not discussed here.

sector has absorbed a similar quantity of sunflower oil, half of it being imported. These products compensated in the food sector for rapeseed and soybean oil transferred to the industrial uses.

For instance, we analysed whether the food consumption varied for the different oils types in the EU, or whether difference of price between rapeseed and palm oil could explain some changes in trade patterns. On the basis of this analysis, we concluded that some substitution of vegetable oil was observed in food demand but was overall relatively limited compared to the industrial demand. Therefore, a relatively low elasticity of substitution should be used in the case of the EU.

On the model side, in order to implement this limited substitution effect, we created an aggregated vegetable oil food item, in which the fluctuation of the different oil shares is relatively constrained. For this purpose, the objective function of GLOBIOM was modified to include some non-linear costs associated to the change in composition of the vegetable oil aggregate. In the version of IFPRI-MIRAGE used in Al-Riffai, Dimaranan & Laborde (2010), an elasticity of substitution of 2 was used in the different regions for substitution of vegetable oils and a trade Armington elasticity of 10.[63] When prices increase, both rapeseed imports and other oil demand react to compensate the shock. An analysis of historical development in the EU for oilseed markets shows that rapeseed oil and rapeseed imports increased five times more than food demand for rapeseed oil decreased, between 2000 and 2012 (Figure 10). In MIRAGE, the contribution of rapeseed demand change through substitution was found contributing more than two times more to the new demand for rapeseed oil.[64]

Uncertainty on the right substitution level is therefore key for the magnitude of responses on the vegetable oil market. As starting point, we will consider in our analysis an elasticity value of 0.8 for the EU, smaller than MIRAGE. The value of this elasticity will be part of the sensitivity analysis. However, for other regions like the US, observed recent changes in consumption of soybeans suggest that higher substitution is possible within the food sector. We will therefore keep a value of 2 in the USA. These values will be varied through the Monte-Carlo, in order to capture effect of having lower or higher substitution effects.

3. Implication for model results

Vegetable oil markets were already connected in GLOBIOM through demand for biofuel use. The present improvement in the model will introduce some possibility of substitution on the food market side, but with a more limited possibilities, reflecting the stickiness observed in the past time series, in particular in the case of the EU. Trade should therefore remain for the EU the most important driver of propagation of demand shock. In the rest of the world, markets will also be connected through the industrial demand and through the food market, in particular for some regions like the US showing larger substitution patterns.

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[63] Data retrieved on ec.europa.eu/energy/renewables/studies/doc/land_use_change/iluc_report_annex_1.xls

[64] Analysis of MIRAGE results from Al Riffai et al., 2010 suggests that for a shock of 324,000 tonnes of rapeseed oil to the biofuel, the EU food market provides 65,000 tonnes, ie 20%. Figure 7 shows that as the rapeseed consumption increased by 6 million tons in the EU in the 2000s, rapeseed decreased by 500,000 tons, ie the food sector did not contribute more than 8%, ie 2.5 times less. See Table 19 for calculation details.

Table 19. Analysis of vegetable oil final consumption in the EU in the biofuel scenario from Al-Riffai et al., 2010 (using appendix results tables)

	Feedstock use for biofuels (1000 t) **(Al-Riffai et al., 2010, Table S6b)**			Food consumption change (%) (Al-Riffai et al., 2010, Table S7)	Food consumption level (1000 t) (USDA, 2010)	Food consumption change (1000 t)	Ratio food change / feedstock demand for biofuel
	Baseline 2020	Scenario 2020	Difference				
Palm oil	824	1008	185	1.08%	2750	30	0.161
Rapeseed oil	4997	5320	324	-2.39%	2733	-65	-0.202
Soybean oil	2978	3441	463	-0.41%	1290	-5	-0.011
Sunflower oil	430	511	81	0.30%	3191	10	0.121

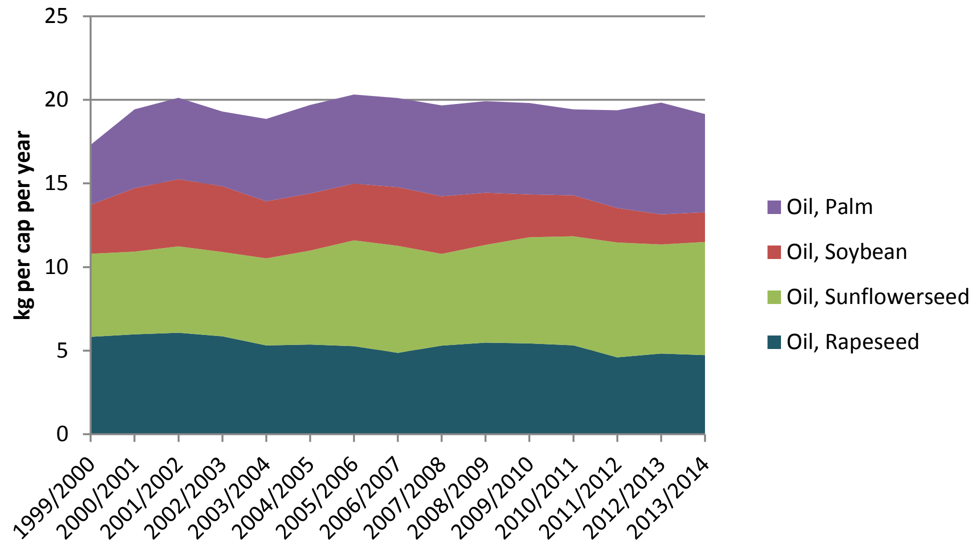

Figure 7 Consumption per capita of vegetable oil by EU consumer according to USDA PSD statistics

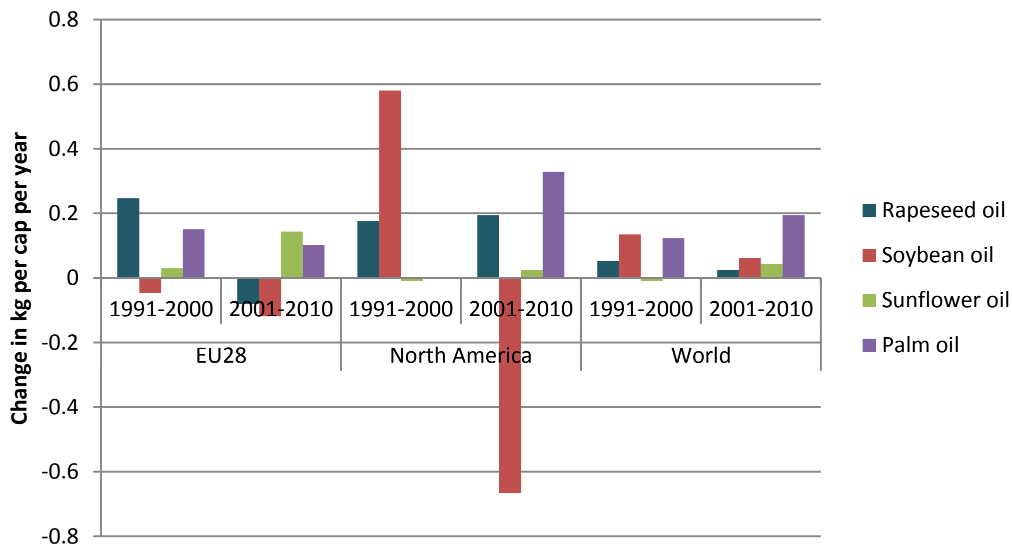

Figure 8 Change in consumption per capita of vegetal oil in EU28, North America and World on two periods 1991-2000 and 2001-2010. Rate of change are obtained by regression of consumption per capita on each period using USDA PSD data on food use.

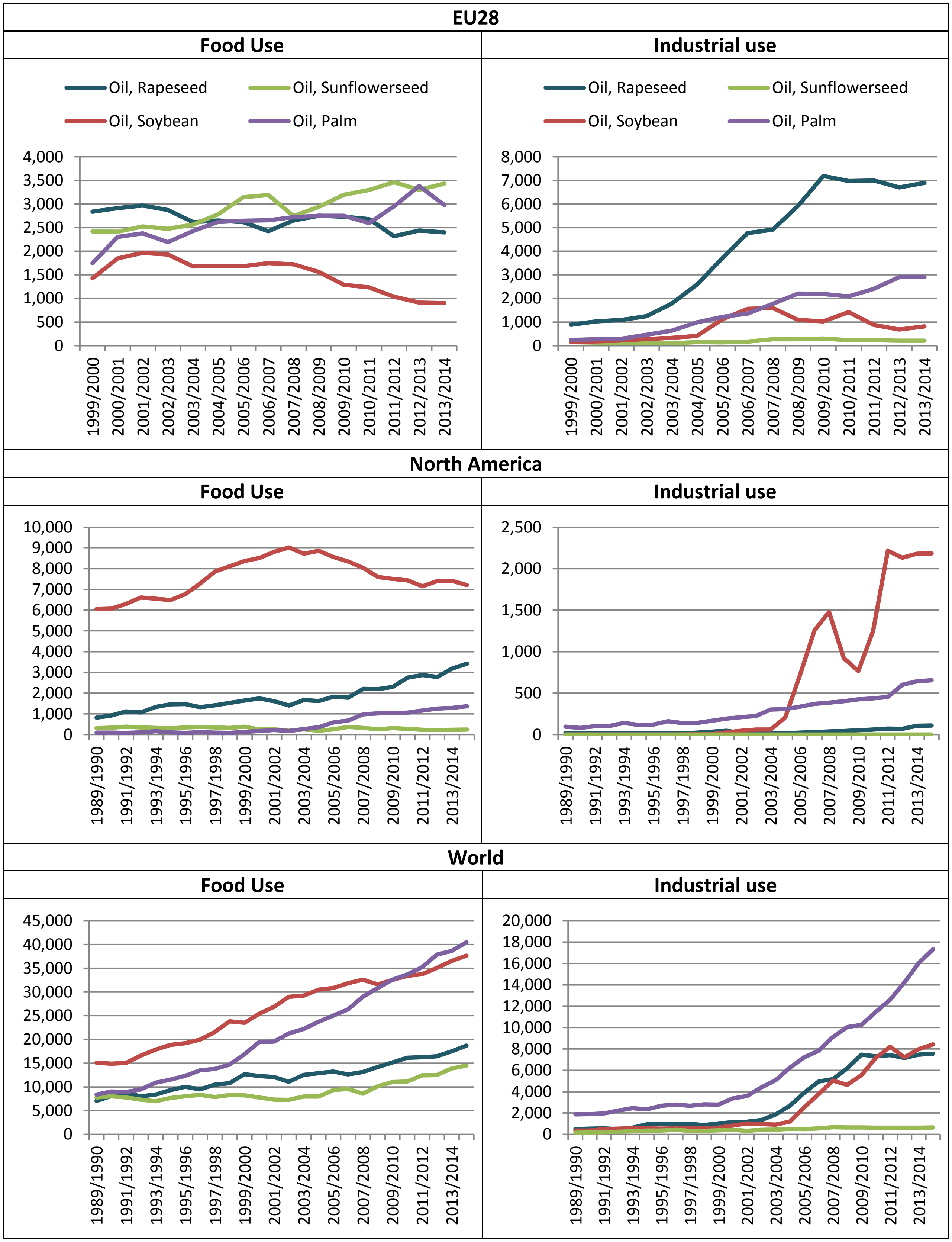

Figure 9. Vegetable oil consumption in the food and the industrial sectors (including biofuels) between 1990 and 2014 according to USDA PSD database. (1000 tonnes)

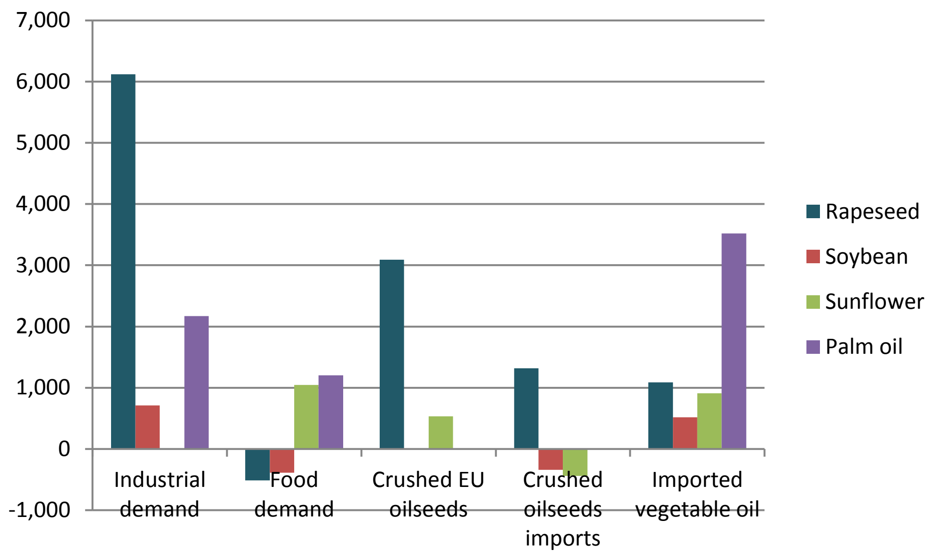

Figure 10. Changes in EU market between 2000 and 2012 for four vegetable oil types (1000 tonnes, source: USDA)

Improvement 24+25: Introducing Argentina, Indonesia, Malaysia and Ukraine in GLOBIOM

1. Motivation for improvements

Argentina, Indonesia and Malaysia are important player on the international biofuel market today: Argentina is a major producer of soybeans and Indonesia and Malaysia concentrate a large majority of the palm oil production. Ukraine could play an important role in the future as a supplier of agricultural products to the EU market, in particular cereals but also rapeseed and sunflower. In order to better represent how these countries can interact with the international markets, it was recommended for the purpose of this project to single them out in order to more precisely trace their trade and how their production level can influence land use patterns, in response to policy scenarios on biofuels.

2. Methodological approach

Most model parameters for the parameterization of the supply side in GLOBIOM (land use, crop area and yield, animal distribution and systems, etc.) are provided at the Simulation Unit level and sourced from biophysical models and downscaling of some national datasets. For these data, input were reprocessed and made compatible with the new regional levels. The demand side had to be disaggregated, using data on quantities and prices from FAOSTAT. Argentina was separated out from previous “Rest of South America” region; Indonesia and Malaysia – previously Rest of South East Asia and Ukraine – previously Former USSR we also singled out. New price elasticities were sourced from USDA. Bilateral trade flows were recalibrated for all the new regions based on COMTRADE and tariffs from MAcMap, following the methodology used so far in GLOBIOM.

Although most data on the supply side were already available through the different input datasets, some adjustments had to be performed to represent adequately some production patterns of the new regions. Indeed, 2000 data from SPAM (Spatial Production Allocation Model), used for the calibration of the initial crop areas, was not available for rape and sunflower in some regions, in particular Ukraine. Some special treatment had to be applied to allocate these crops spatially. Initial crop areas were distributed across Simulation Units und management system using the SPAM information on potential “pre-crops” in the rotations of sunflower and rapeseed (typically barley, corn, or wheat).

3. Implication for model results

The new regional mapping of GLOBIOM now gives access to more precise characterization of trade and uses of products in the new regions. Demand quantities, trade flows, prices can now be reported for these regions separately. On the supply side, land use change patterns can be more precisely connected to trade, as market resolution has been increased in the new areas of interest and localization of production changes is now more precisely assessed.

Table 20. List of regions newly represented in GLOBIOM for the ILUC study (new countries in bold)

GLOBIOM region	Definition
ANZ	Australia, New Zealand
Argentina	**Argentina**
Brazil	Brazil
Canada	Canada
China	China
Congo Basin:	Cameroon, Central African Republic, Congo Republic, Democratic Republic of Congo, Equatorial Guinea, Gabon
EU28, each country is treated as separate region	Baltic: Estonia, Latvia, Lithuania East: Bulgaria, Croatia, Czech Republic, Hungary, Poland, Romania, Slovakia, Slovenia, Central: Austria, Belgium, Germany, France, Luxembourg, Netherlands North: Denmark, Finland, Ireland, Sweden, United Kingdom South: Cyprus, Greece, Italy, Malta, Portugal, Spain
Former USSR	Armenia, Azerbaijan, Belarus, Georgia, Kazakhstan, Kyrgyzstan, Moldova, Russian Federation, Tajikistan, Turkmenistan, Uzbekistan
India	India
Indonesia	**Indonesia**
Japan	Japan
Malaysia	**Malaysia**
Mexico	Mexico
Middle East and North Africa	Algeria, Bahrain, Egypt, Iran, Iraq, Israel, Jordan, Kuwait, Lebanon, Libya, Morocco, Oman, Qatar, Saudi Arabia, Syria, Tunisia, United Arab Emirates, Yemen
Pacific Islands	Fiji Islands, Kiribati, Papua New Guinea, Samoa, Solomon Islands, Tonga, Vanuatu
RCAM	Bahamas, Barbados, Belize, Bermuda, Costa Rica, Cuba, Dominica, Dominican Republic, El Salvador, Grenada, Guatemala, Haiti, Honduras, Jamaica, Nicaragua, Netherland Antilles, Panama, St Lucia, St Vincent, Trinidad and Tobago
RCEU	Albania, Bosnia and Herzegovina, Macedonia, Serbia-Montenegro
ROWE	Gibraltar, Iceland, Norway, Switzerland
RSAM	Bolivia, Chile, Colombia, Ecuador, Guyana, Paraguay, Peru, Suriname, Uruguay, Venezuela
RSAS	Afghanistan, Bangladesh, Bhutan, Maldives, Nepal, Pakistan, Sri Lanka
RSEA OPA	Brunei Daressalam, Singapore, Myanmar, Philippines, Thailand
RSEA PAC	Cambodia, Korea DPR, Laos, Mongolia, Viet Nam
South Africa	South Africa
South Korea	South Korea
Eastern Africa	Burundi, Ethiopia, Kenya, Rwanda, Tanzania, Uganda
Southern Africa	Angola, Botswana, Comoros, Lesotho, Madagascar, Malawi, Mauritius, Mozambique, Namibia, Reunion, Swaziland, Zambia, Zimbabwe
Western Africa	Benin, Burkina Faso, Cape Verde, Chad, Coted Ivoire, Djibouti, Eritrea, Gambia, Ghana, Guinea, Guinea Bissau, Liberia, Mali, Mauritania, Niger, Nigeria, Senegal, SierraLeone, Somalia, Sudan, Togo
Turkey	Turkey
Ukraine	**Ukraine**
United States of America	United States of America

Improvement 27: Representing expansion into unused land

1. Motivation for improvements

Agriculture represented 44% of land use in the European Union in 2000. In GLOBIOM, this land is represented through various land categories: "Cropland" corresponding to the share of this land used to produce the crops represented in GLOBIOM; "Pasture" corresponding to areas used for ruminant grazing. A part of this land is also occupied by other agricultural activities that are not represented for the moment in GLOBIOM (e.g. vegetables, vineyards, orchards). These are identified as "other agricultural land" and this land is kept fixed in the model. Beside these managed land, some unmanaged land are also input to the model: "natural forest", "wetlands", and "other natural land", that contain all remaining types of fertile areas.

Unused agricultural land falls in three categories:

- set-aside land is represented in the EU directly in the crop rotation as a crop management option subject to profit maximization,
- other fallow land declared as cropland is part of the "Other agricultural land category",
- abandoned land no longer declared as cropland is accounted outside of the agricultural land under the category "other natural land".

The extent of "other natural land" in GLOBIOM is usually much greater than the "other agricultural land" category. Therefore, even if "other agricultural land" is fixed, the potential agricultural land expansion is large. However, depending on the location, "forest" land can also be used in the expansion. In the standard version of GLOBIOM, both "forest" and "other natural land" are used when agricultural land expand, with proportions determined by the calibration parameters and based on observations of past land use changes.

It has been argued that the potential contribution of unused land in Europe has been underestimated in past assessments of land use change dynamics and should be better represented in GLOBIOM. For that reason, it was decided to perform some scenarios where the access to unused land would be facilitated in the EU as well as in Ukraine, a large potential supplier of agricultural products to Europe which already today provides significant quantities of biofuel feedstock to the EU.

2. Methodological approach

This improvement is performed as a new scenario (scenario C) with a change of parameterisation of the model. For this scenario, we model possibilities of expansion through the third unused agricultural land category detailed above. Access cost to "other natural land" is therefore reduced for this land use type in all countries of the European Union as well as for Ukraine.

For all EU regions, this land use category is large. In the case of Ukraine, input data were also improved to better account for recent assessment on abandoned land based from Alcantara et al. (2013) who find that 9.2 Mha of farmland is currently abandoned in Ukraine. This land area was therefore classified

under “Other natural vegetation” instead of “Other agricultural land”, homogeneously across the country.

3. Implication for model results

Scenario C has been especially designed to represent the effect of improved access to unused agricultural land. The effect of the change in access cost in scenario C will be an increased share of agricultural expansion in the EU and Ukraine, with greater use of the “other natural vegetation” land use category.

Table 21. Land use in the EU and in Ukraine in the GLOBIOM nomenclature in 2000 (1000 ha)

Country	Cropland	Pasture	Other agricultural land	Forest	Other natural vegetation	Wetlands
Austria	1,319	1,705	97	3,828	494	16
Belgium	809	675	104	653	191	10
Bulgaria	2,570	1,511	192	3,507	2,303	11
Croatia	852	1,609	488	2,129	301	
Cyprus	71	46	195	148	410	
CzechRep	2,846	631	137	2,569	1,170	8
Denmark	2,264	107	80	526	906	62
Estonia	610	48	7	2,170	1,076	188
Finland	1,712	460	28	21,953	4,893	867
France	17,989	7,637	736	13,787	9,783	181
Germany	11,505	4,360	851	10,751	4,784	178
Greece	2,173	1,836	187	1,188	6,054	39
Hungary	3,229	1,025	366	1,661	2,093	63
Ireland	264	2,778	126	461	2,039	1,002
Italy	7,091	3,328	641	7,452	6,553	58
Latvia	940	231	73	3,148	1,661	149
Lithuania	1,490	408	405	1,924	1,862	55
Luxembourg	38	46	17	89	48	
Malta	5	0	14		6	
Netherlands	851	932	149	345	710	31
Poland	12,067	3,774	1,072	8,879	3,752	132
Portugal	1,263	1,102	104	3,825	1,491	25
Romania	6,815	4,836	354	6,698	2,591	247
Slovakia	1,355	494	78	1,980	647	5
Slovenia	154	276	11	1,188	284	3
Spain	11,185	6,495	603	12,580	13,514	93
Sweden	2,734	206	120	24,077	8,911	3,514
UK	5,539	7,950	275	2,603	5,579	438
Ukraine	14,419	17,392	7,415	8,893	10,361	259

Improvement 34+35: Improving biofuel supply chains conversion coefficients

1. Motivation for improvements

The past assessment of ILUC has raised some concerns about the conversion coefficients to be used at different stages of the processing of agricultural materials into biofuels. GLOBIOM offers an explicit representation of conversion technologies and it was decided to document the current conversion assumptions to give opportunity to agriculture and industry stakeholder to comment on the assumptions for i) oilseeds crushing supply chains (improvement 34), ii) bioethanol and biodiesel transformation chains (improvement 35).

2. Methodological approach

A data document was compiled containing most important assumptions for GLOBIOM supply chains. This document will be made public once all input from stakeholders will have been reviewed and initial GLOBIOM assumptions improved when relevant. A list of input and comments on assumptions received during the consultation period is provided in Table 22. Corrections performed are also reported in that table when they were considered relevant. In some cases, reported issues led to direct adjustments in the initial GLOBIOM data (e.g. on biofuel supply chains), or only in adjustment of coefficients along the baseline (e.g. for crushing rates that vary over time).

Comments received were in particular on the following topics:

- Crushing rate values in Europe and rest of the world (FEDIOL)
- Conversion efficiency for corn and maize (Epure)
- Conversion efficiency for sugar beet and sugar cane (CGB)
- Final use of vegetable oils (FEDIOL)
- Production level of sugar beet (CGB)
- Conversion efficiency of sugar beet (CGB)
- Conversion efficiency of sugar cane (CGB)

3. Implication for model results

Adjustments of supply chains parameters will allow a more precise description of land use requirements associated to the different feedstocks, and improve the assessment of indirect land use change effects.

Table 22. List of comments received on supply chains specifications in GLOBIOM and actions taken

Source	Comment	Comment	Correction necessary
FEDIOL (Jan 14)	Crushing rates for the EU	We compared EU crushing rates with our numbers. Values are very close to our estimates but with slightly higher moisture content (2-3%). These estimates were updated in contribution from FEDIOL from May 2014.	Yes
FEDIOL (Jan 14)	Split end-use for the EU in 2011-2012	We compared these data with our FAO and EUROSTAT sources and we found some consistent shares of uses overall. The data will be compared again after the baseline has been produced.	Yes
FEDIOL (May 2014)	Crushing rates for major producing countries	EU crushing coefficients were found very close for rapeseed and sunflower, and identical for soybeans to GLOBIOM assumptions. We adjusted our coefficient to FEDIOL values for the year 2010 and following. For other countries than EU, our estimates were also very close but adjusted as well.	Yes
EBB (April 2014)	Production of glycerin	Due to its too loose connection to land use change dynamics, it has been decided not to change the supply chain representation in the model to introduce glycerin. Although glycerin can impact the life cycle assessment of biofuels, only land use change emissions are the focus of the study.	No
Epure (June 2014)	Corn ethanol conversion coefficient in the EU: 400-427 l/t	Estimates provided are 6 to 13% above JRC values. However, US EPA usually uses a value of around 417 l/ton. This value will be used as a reference unless more specific data on the EU are provided by JRC.	Yes
Epure (June 2014)	Wheat ethanol conversion coefficient in the EU: 0.29-0.295 t/t.	The value proposed by Epure are 1-3% above Biograce values. Except if authoritative reference is provided, we assumed Biograce default was acceptable.	No
Epure (June 2014)	Wheat DDGS and corn DDGS output should be comparable: 0.29-0.32 t/t crop at 10% mc	Biograce value for corn was updated with latest version of Wells to Tank analysis from JRC (2014 version 4a). Yields are now for DDGS of 0.31 tons DDGS 0% mc for wheat and corn at 10% mc.	Yes
Epure (June 2014)	Wheat DDGS conversion ratio is too low.	The numbers provided is 0.294 with 0% mc, which is equivalent to 0.326 with 10% mc. The value from Biograce for wheat DDGS seems therefore in line with Epure input.	No
Epure (June 2014)	Sugar processing coproducts vinasse and carbonate lime are not represented.	We did not change the supply chains to add products that do not interact with land use change dynamics, because the project only looks at LUC emissions.	No
Epure (June 2014) / CGB (March 2014)	Sugar content assumed for sugar beet is too low: average of 17.6% should be assumed for past 5 years.	We analysed in details statistics received from CGB. We concluded that in order to best reflect the heterogeneity of sugar content across production of member states, the most consistent approach in GLOBIOM was to recalculate all yield values for beet at 16% sugar content. We will for this use CIBE information on yield in ton sugar / ha per Member State and divide by 0.16 to obtain beet yield and production.	Yes
Epure (June 2014)	Sugar production volume are not correct at EU MS level	As explained above, our production statistics will be updated taking harmonizing yield at 16% sugar content.	Yes
CGB (March 2014)	Average yield values should be used rather than point estimates	This is currently the way it is done in GLOBIOM: we use a 3 year average on the period 1999-2001 for the base year yield level.	No
CGB (March 2014)	Yield improvement should be taken into account, sugar beet had a strong yield improvement over past years.	We take yield improvement into account in our baseline; our yield improvement assumptions will be calibrated on CAPRI model projections used by DG Agriculture. If longer time series on sugar yield per ha are provided, we can also introduce a trend on sugar content in beet.	Yes
CGB (March 2014)	Sugar cane area not harvest should be accounted to reflect correct apparent	After check, area reported by FAOSTAT as “harvested” for Brazil correspond to the total area under sugar cane.	No

	sugar cane yield (different from field yield)		
CGB (March 2014)	Sugar beet sugar content for ethanol production should take into account the fact that EU ethanol producers have higher sugar content in beet than average EU.	We will adjust our production statistics to reflect the actual average sugar content of each Member state and correct production for 16% sugar equivalent. Therefore, the conversion of ethanol will be the same for all EU beet based on this 16% and no further adjustment will be needed. The yield will be equal to the actual 18.2% once ethanol production will be adequately allocated across member states.	No
CGB (March 2014)	Sugar cane conversion rate should be checked to reflect dehydrated conversion efficiency instead of hydrated one.	We checked the value used in our tables based on Biograce/JRC of 1.77 GJ / ton sugar cane. This corresponds after conversion to 83.6 liter ethanol / ton sugar cane. This is slightly lower than the 86.3 liter reported by CGB but considering the past average sugar content over 5 years was found to be 138 kg / ton sugar cane, this seems consistent (CGB assumed for their calculation 142 kg/ton sugar cane).	No
CGB (March 2014)	Sugar beet conversion factor from JRC should be applied to beet at 16% sugar content, not to actual yield.	As explained above, we will indeed adjust our production and yield values in GLOBIOM to reflect production of beet at 16% sugar content and not at actual content. The JRC conversion factor will therefore remain relevant.	Yes

Appendix: Bioenergy transformation pathways specifications

The mass balance of each biofuel pathway is given below as it is currently implemented in the model. For most biofuel pathways, the total feedstock to fuel conversion is described as one step; for some pathways, the conversion is described in two steps via an intermediate product (e.g. vegetable oil). Conversion coefficients are applied worldwide, except where indicated otherwise. The use or co-production of energy is not described because costs are inherently included in the transformation costs.

Table 23 Energy content of various biofuel types

Fuel type	Energy content
Bioethanol [a)]	26.81 MJ/kg
Biodiesel (FAME) [a)]	37.2 MJ/kg
Biodiesel (HVO) [a)]	44 MJ/kg
Biodiesel (Fischer Tropsch) [a)]	43.92 MJ/kg
Butanol [b)]	33 MJ/kg
Methanol [b)]	20 MJ/kg
Bio DME [b)]	28 MJ/kg
Methane (upgraded biogas) [b)]	50 MJ/kg or about 33 MJ/m3

a) Biograce (2014)

b) Renewable Energy Directive 2009/28/EC.

1. Ethanol conventional

Corn ethanol

Product	Region	Unit	Input	Output
Corn		tonne (15% mc)	**-1**	
Ethanol	USA [a)]	GJ		**8.68**
		tonne (0% mc)		**0.318**
	EU & ROW [b)]	GJ		**8.72**
		tonne (0% mc)		**0.319**
Corn DDGS	USA [a)]	tonne (0% mc)		**0.304**
		GJ		**5.42**
	EU & ROW [c)]	tonne (0% mc)		**0.295**
		GJ		**5.26**

a) 2.76 gallon ethanol and 17 lbs of dried distillers grains per bushel corn (EPA, 2010), with LHV corn at 18.5 MJ/kg at 0% mc, LHV ethanol at 26.81 MJ/kg at 0% mc and LHV DDGS at 16.0 MJ/kg at 10% mc.
b) Edwards et al. (2004). Revision V4 (2014). Pathway "Production of Ethanol from Corn (Community produced) (steam from natural gas CHP)". Overall yield is 0.6032 MJ ethanol/MJ corn, with LHV corn at 17 MJ/kg at 0% mc and LHV ethanol at 26.81 MJ/kg at 0% mc.
c) Ibid, Yield of DDGS is 1.392 tonne DDGS/tonne ethanol, with DDGS at 10% mc. LHV DDGS is 16.0 MJ/kg at 10% mc. Biograce does not make distinction between the energy content of corn DDGS and wheat DDGS. Note that Globiom will not use LHV for DDGS but more metabolizable energy by animal.

Wheat ethanol

Product	Region	Unit	Input	Output
Wheat		tonne (15% mc)	**-1**	
Ethanol [a)]	Global	GJ		**7.68**
		tonne (0% mc)		**0.286**
Wheat DDGS [b)]	Global	tonne (0% mc)		**0.294**
		GJ		**5.22**

a) Biograce (2014). Pathway "Production of Ethanol from Wheat (steam from natural gas CHP)". Overall yield is 0.5313 MJ ethanol/MJ wheat, with LHV wheat at 17.0 MJ/kg at 0% mc and LHV ethanol at 26.81 MJ/kg at 0% mc
b) Ibid. Yield of DDGS is 1.14 tonne DDGS/tonne ethanol, with DDGS at 10% mc. LHV DDGS at 16.0 MJ/kg at 10% mc.

Rye ethanol

Since the starch content of rye is approximately the same as for wheat, the same conversion efficiencies and costs are assumed. Wheat and rye are processed in the same ethanol facility, with feedstock mix depending on availability and cost.

Product	Region	Unit	Input	Output
Rye		tonne (15% mc)	**-1**	
Ethanol	Global	GJ		**7.68**
		tonne (0% mc)		**0.286**
Rye DDGS	Global	tonne (0% mc)		**0.294**
		GJ		**5.22**

Sugar beet ethanol

Product	Region	Unit	Input	Output
Sugar beet		tonne (76% mc)	**-1**	
Ethanol [a)]	Global	GJ		**2.13**
		tonne		**0.079**
Sugar fibre [b)]	Global	tonne (0% mc)		**0.055**
		GJ		**0.857**

a) Biograce (2014). Pathway "Production of Ethanol from Sugarbeet (steam from NG boiler). Overall yield is 0.5436 MJ ethanol/MJ sugar beet, with LHV sugar beet at 16.3 MJ/kg at 0% mc and LHV ethanol at 26.81 MJ/kg at 0% mc.

b) Ibid. Yield of co-product is 0.219 MJ sugar beet pulp/MJ sugar beet, with LHV sugar beet pulp at 15.6 MJ/kg at 0% mc.

Sugar cane ethanol

Product	Region	Unit	Input	Output
Sugar cane		tonne (75% mc)	**-1**	
Ethanol [a)]	Global	GJ		**1.77**
		tonne (0% mc)		**0.066**
Bagasse [b)]	Global	kWh		**N/A**

a) Biograce (2014). Pathway "Production of Ethanol from Sugarcane". Overall yield is 0.3607 MJ ethanol/MJ sugar cane, with LHV sugar cane at 19.6 MJ/kg at 0% mc and LHV ethanol at 26.81 MJ/kg at 0% mc.

b) Electricity cogeneration is not explicitly represented in GLOBIOM for sugar cane processing but accounted for through the absence of energy cost for production.

2. Biodiesel conventional

Oilseed crushing

The crushing ratios currently used in the model are derived from data provided by national statistic offices and accessible through Eurostat or FAOSTAT. Crushing rates and crushing efficiency are then reproduced in the model as they appear. Within EU, national statistics display some variations that may not necessarily correspond to differences in technologies used but most likely in heterogeneity in crop processed. For that reason, we only use one average EU crushing rate.

The crushing ratios for oil used here should be interpreted as seed to crude oil crushing ratios. Conversion to biodiesel later requires a vegetable oil refining stage that is accounted for separately. Cake and oil do not sum to 100% due to seed moisture extraction and in some cases additional losses.

Table 24. Crushing ratio oilseeds (1999-2001 average) a)

	Rapeseed			Soybean			Sunflower		
	Cake	Oil	Total	Cake	Oil	Total	SunC	SunO	Total
EU28 b)	**57**	**42**	**99**	**80**	**18**	**98**	**56**	**43**	**99**
Other regions									
Brazil				80	20	100			
Canada c)	56	42	98	78	18	96			
China	62	36	98	82	18	100	50	35	85
Former USSR							47	46	93
India	60	35	95	80	18	98			
Japan	57	42	99	77	19	96			
Mexico				80	15	95			
Middle East & North Africa				80	17	97			
Rest of South America				79	18	97	42	41	83
Rest of South Asia	62	33	95						
South-East Asia				80	18	98			
South Korea				76	18	94			
Turkey							45	38	83
USA				79	19	98			

a) FAOSTAT, UN Food and Agriculture Organization. We report here ratios for crushed quantities higher than 1 million tonnes.
b) EUROSTAT/CAPRI database.
c) For Canada, rapeseed cake and oil mass from crushing in FAOSTAT data exceed mass for processed rapeseed. Meal mass was adjusted based on information on moisture content (12%) from the Canola Council of Canada.

Vegetable oil refining

The use of crude vegetable oil as a feedstock for biodiesel involves a refining stage, leading to some losses. We currently apply 4% by mass loss for all regions and vegetable oil types.

Product	Unit	Input	Output
Crude vegetable oil	tonne	**-1**	
Refined vegetable oil [a)]	tonne		**0.960**

a) Edwards et al. (2004).

Fatty Acid Methyl-Ester processing (FAME)

Product	Region	Unit	Input	Output
Refined oil		tonne (0% mc)	**-1**	
FAME [a)]	Global	GJ		**36.6**
		tonne (0% mc)		**0.983**

a) Biograce (2014). Pathway "Production of FAME from Rapeseed (steam from natural gas boiler)". Yield is 0.9936 MJ FAME/MJ refined oil, with LHV FAME at 37.2 MJ/kg at 0% mc. The LHV refined oil is not given by Biograce, we assume it is similar to that of soybean and palm oil at 36.8 MJ/kg at 0% mc. Furthermore, refined glycerol is coproduced at 105.6 kg / tonne FAME. The co-production of glycerol is accounted for in the processing costs. The Globiom model does not take into account trickle down effects of glycerol, even though it can be used to produce biofuels, such as biomethanol (production of fuels on basis of residues is separately accounted for).

Hydrotreated vegetable oil (HVO)

Product	Region	Unit	Input	Output
Vegetable oil		tonne (0% mc)	**1**	
HVO [a)]	Global	GJ		**34.8**
		tonne (0% mc)		**0.791**

a) Biograce (2014). Pathway "Production of HVO from Rapeseed (steam from natural gas boiler)". Yield is 0.967 MJ HVO/MJ oil (not refined), with LHV HVO at 44.0 MJ/kg at 0% mc. Assume that LHV vegetable oil is 36.0 MJ/kg at 0% mc. Biograce does not specify co-product, although other sources mention gasoline and propane as side products.

3. Biogas

Biogas from maize silage.

Product	Region	Unit	Input	Output
Maize silage		tonne (0% mc)	**1**	
Biogas [a)]	Global	GJ		**9.9**
		tonne (0% mc)		**0.198**

a) Typical yield from (IEA, 2011) slide 5, biogas from whole crop maize is 178 – 400 m3 methane per tonne dry matter (mainly depending on the feedstock quality and retention time), so use 300 m3 as average value, with lower heating value methane at 33 MJ/m3. However, about 25% of energy produced is used to drive the complete process (digester and upgrading). Density methane is 0.66 kg/m3.

4. Advanced biofuels

Cellulosic ethanol from woody biomass (fermentation)

This pathway should be seen as a container of future technologies producing alcohols from lignocellulosic biomass.

Product	Region	Unit	Input	Output
Wood		tonne (0% mc)	**-1**	
Ethanol [a)]		GJ		**9.32**
		tonne (0% mc)		**0.348**

a) IRENA (2013) Table 4.2: The average yield is 440 liters of ethanol per tonne (0% mc) wood. Assume LHV ethanol at 26.81 MJ/kg at 0% mc as in other tables above, and a density of 0.79 kg/litre.

Diesel from woody biomass (gasification and synthesis)

This pathway should be seen as a container of future technologies producing diesel-like fuels from lignocellulosic biomass.

Product	Region	Unit	Input	Output
Wood		tonne (0% mc)	**-1**	
FT Diesel [a)]		GJ		**9.37**
		tonne (0% mc)		**0.213**

a) Dimitriou (2013) compares FT diesel production via entrained flow gasification and circulating fluidised bed gasification and finds comparable outcomes. We have used the parameters for the CFB pathway here: At an input of 120 tonne/hr wet biomass (30% mc and LHV 13,056 MJ/kg), thus 84 tonne at 0% mc, the output is 17.93 tonne/hr FT diesel at 43.92 MJ/kg.

References

Agus, F., Handayani, E., van Noordwijk, M., Idris, K., & Sabiham, S. (2010). Root respiration interferes with peat CO2 emission measurement. In *19th World Congress Congress of Soil Science, Soil Solutions for a Changing World* (pp. 50–53). Brisbane, Australia.

Agus, F., Henson, I., Sahardjo, B. H., Harris, N., van Noordwijk, M., & Killeen, T. (2013). *Review of emission factors for assessment of CO2 emission from land use change to oil palm in Southeast Asia* (pp. 7–28). Kuala Lumpur.

Alcantara, C., Kuemmerle, T., Baumann, M., Bragina, E. V, Griffiths, P., Hostert, P., … Radeloff, V. C. (2013). Mapping the extent of abandoned farmland in Central and Eastern Europe using MODIS time series satellite data. *Environmental Research Letters*, *8*(3), 35035. Retrieved from http://stacks.iop.org/1748-9326/8/i=3/a=035035

Al-Riffai, P., Dimaranan, B., & Laborde, D. (2010). *Global Trade and Environmental Impact Study of the EU Biofuels Mandate*. *Final Report for the Directorate General for Trade of the European Commission*.

Anderson, P. V., Kerr, B. J., Weber, T. E., Ziemer, C. J., & Shurson, G. C. (2012). Determination and prediction of digestible and metabolizable energy from chemical analysis of corn coproducts fed to finishing pigs. *Journal of Animal Science*, *90*, 1242–1254.

Asikainen, A., Liiri, H., Peltola, S., Karjalainen, T., & Laitila, J. (2008). Forest energy potential in Europe (EU 27). Metla, Finnish Forest Research Institute. Retrieved from http://www.metla.fi/julkaisut/workingpapers/2008/mwp069-en.htm

Biograce. (2014). *Biograce version 4c (April 2014)*.

BIOMASS FUTURES. (2012). *Atlas of EU biomass potentials. Deliverable 3.3: Spatially detailed and quantified overview of EU biomass potential taking into account the main criteria determining biomass availability from different sources*. Retrieved from http://www.biomassfutures.eu/public_docs/final_deliverables/WP3/D3.3 Atlas of technical and economic biomass potential.pdf

Blanco-Canqui, H., & Lal, R. (2007). Soil and crop response to harvesting corn residues for biofuel production. *Geoderma*, *141*, 355–362. doi:10.1016/j.geoderma.2007.06.012

Carlson, K. M., Curran, L. M., Asner, G. P., Pittman, A. M., Trigg, S. N., & Marion Adeney, J. (2012). Carbon emissions from forest conversion by Kalimantan oil palm plantations. *Nature Climate Change*, *3*(3), 283–287. doi:10.1038/nclimate1702

Carré., F., Hiederer, R., Blujdea, V., & Koeble, R. (2010). *Background Guide for the Calculation of Land Carbon Stocks in the Biofuels Sustainability Scheme Drawing on the 2006 IPCC Guidelines for National Greenhouse Gas Inventories.* Luxembourg: Office for Official Publications of the European Communities.

Comeau, L., Hergoualc'h, K., Smith, J., & Verchot, L. (2013). *Conversion of intact peat swamp forest to oil palm plantation: effects on soil CO 2 fluxes in Jambi* (No. 110). Bogor, Indonesia.

Couwenberg, J. (2009). *Emission factors for managed peat soils: an analysis of IPCC default values.* Wageningen.

Couwenberg, J., Dommain, R., & Joosten, H. (2010). Greenhouse gas fluxes from tropical peatlands in south-east Asia. *Global Change Biology*, *16*(6), 1715–1732. doi:10.1111/j.1365-2486.2009.02016.x

Couwenberg, J., & Hooijer, A. (2013). Towards robust subsidence-based soil carbon emission factors for peat soils in south-east Asia, with special reference to oil palm plantations. *Mires & Peat*, *12*, 1–13.

Dariah, A., Marwanto, S., & Agus, F. (2013). Root- and peat-based CO2 emissions from oil palm plantations. *Mitigation and Adaptation Strategies for Global Change*, (12).

Derpsch, R., Friedrich, T., Kassam, A., & Hongwen, L. (2010). Current status of adoption of no-till farming in the world and some of its main benefits. *International Journal of Agriculture and Biological Engineering*, *3*(1), 1–26. doi:10.3965/j.issn.1934-6344.2010.01.0-0

Dimitriou, I. (2013). *Techno-economic assessment and uncertainty analysis of thermochemical processes for second generation biofuels*. *Aston University*.

Ecofys. (2013). *Low ILUC potential of wastes and residues for biofuels. Straw, forestry residues, UCO, corn cobs*. Retrieved from http://www.ecofys.com/files/files/ecofys-2013-low-iluc-potential-of-wastes-and-residues.pdf

Edwards, R., Mahieu, V., Griesemann, J. C., Larive, J. F., & Rickeard, D. J. (2004). Well-to-wheels Analysis of Future Automotive Fuels and Powertrains in the {E}uropean Context. *SAE Transactions*, *113*(4), 1072–1084.

EPA. (2010). *Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis*. Retrieved from http://www.epa.gov/otaq/fuels/renewablefuels/regulations.htm

EPA. (2012). Notice of Data Availability Concerning Fuels Produced from Palm Oil Under the RFS Program. *Federal Register*, *77*(18), 4300–4318.

FAO. (2010). *Global Forest Resources Assessment*. Retrieved from http://www.fao.org/forestry/fra/fra2010/en/

FAO. (2014). *FAOSTAT Gateway*.

FAO, IIASA, ISRIC, ISS-CAS, & JRC. (2012). *Harmonized World Soil Database (version 1.2)*. Retrieved from http://webarchive.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/

Gunarso, P., Hartoyo, M. E., Agus, F., & Killeen, T. (2013). *Oil palm and land use change in Indonesia, Malaysia and Papua New Guinea* (pp. 29–64). Singapore.

Harris, N., Killeen, T., Brown, K., Netzer, M., & Gunarso, P. (2013). *Pojections of oil palm expansion in Indonesia, Malaysia and Papua New Guinea from 2010 to 2050*. *rspo.org* (pp. 89–112). Jakarta.

Hergoualc'h, K., & Verchot, L. V. (2011). Stocks and fluxes of carbon associated with land use change in Southeast Asian tropical peatlands: A review. *Global Biogeochemical Cycles*, *25*(2), n/a–n/a. doi:10.1029/2009GB003718

Hergoualc'h, K., & Verchot, L. V. (2013). Greenhouse gas emission factors for land use and land-use change in Southeast Asian peatlands. *Mitigation and Adaptation Strategies for Global Change*. doi:10.1007/s11027-013-9511-x

Herrero, M., Havlík, P., Valin, H., Notenbaert, A., Rufino, M. C., Thornton, P. K., … Obersteiner, M. (2013). Biomass use, production, feed efficiencies, and greenhouse gas emissions from global livestock systems. *Proceedings of the National Academy of Sciences of the United States of America*, *110*(52), 20888–93. Retrieved from http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=3876224&tool=pmcentrez&rendertype=abstract

Hirano, T., Segah, H., Harada, T., Limin, S., June, T., Hirata, R., & Osaki, M. (2007). Carbon dioxide balance of a tropical peat swamp forest in Kalimantan, Indonesia. *Global Change Biology*, *13*(2), 412–425. doi:10.1111/j.1365-2486.2006.01301.x

Hirano, T., Segah, H., Kusin, K., Limin, S., Takahashi, H., & Osaki, M. (2012). Effects of disturbances on the carbon balance of tropical peat swamp forests. *Global Change Biology*, *18*(11), 3410–3422. doi:10.1111/j.1365-2486.2012.02793.x

Hoffman, L. A., & Baker, A. (2011). *Estimating the Substitution of Distillers' Grains for Corn and Soybean Meal in the U.S. Feed Complex*.

Hooijer, A., Page, S., Canadell, J. G., Silvius, M., Kwadijk, J., Wösten, H., & Jauhiainen, J. (2010). Current and future CO2 emissions from drained peatlands in Southeast Asia. *Biogeosciences*, *7*(5), 1505–1514. doi:10.5194/bg-7-1505-2010

Hooijer, A., Page, S., Jauhiainen, J., Lee, W. ., Lu, X. X., Idris, A., & Anshari, G. (2012). Subsidence and carbon loss in drained tropical peatlands. *Biogeosciences*, *9*(3), 1053–1071. doi:10.5194/bg-9-1053-2012

Hooijer, A., Silvius, M., Wösten, H., & Page, S. E. (2006). *PEAT-CO2. Assessment of CO2 emissions from drained peatlands in SE Asia* (No. Q3943). Delft, Netherlands.

Husnain, H., Wigena, I. G. P., Dariah, A., Marwanto, S., Setyanto, P., & Agus, F. (2014). CO2 emissions from tropical drained peat in Sumatra, Indonesia. *Mitigation and Adaptation Strategies for Global Change*. doi:10.1007/s11027-014-9550-y

IEA. (2011). *Task 37 Energy from Biogas*.

IPCC. (2006). *IPCC guidelines for national greenhouse gas inventories*. (S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe, Eds.). Institute for Global Environmental Strategies, Hayama, Japan.

IPCC. (2013). Drained Inland Organic Soils. In M. Drösler, A. Freibauer, G. Pan, & L. V. Verchot (Eds.), *2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands* (2013th ed., pp. 1–79). Geneva: IPCC, Switzerland.

IRENA. (2013). *Road transport: the cost of renewable solutions - Preliminary findings*.

Jauhiainen, J., Hooijer, A., & Page, S. E. (2012). Carbon dioxide emissions from an Acacia plantation on peatland in Sumatra, Indonesia. *Biogeosciences*, *9*(2), 617–630. doi:10.5194/bg-9-617-2012

Johnson, J. M.-F., & Barbour, N. W. (2010). Crop yield and greenhouse gas responses to stover harvest on glacial till Mollisol. In *19th World Congress of Soil Science, Soil Solutions for a Changing World*. Brisbane, Australia.

Jones, R. J. A., Hiederer, R., Rusco, E., & Montanarella, L. (2005). Estimating organic carbon in the soils of Europe for policy support. *European Journal of Soil Science*, *56*(5), 655–671. doi:10.1111/j.1365-2389.2005.00728.x

Kalscheur, K. F., Garcia, A. D., Schingoethe, D. J., Royon, F. D., & Hippen, A. R. (2012). Feeding biofuel co-products to dairy cattle. In H. P.S.Makkar (Ed.), *Biofuel co-products as livestock feed - Opportunities and challenges*. Rome: FAO.

Khasanah, N., van Noordwijk, M., Ekadinata, A., Dewi, S., Rahayu, S., Ningsih, H., … Octaviani, R. (2012). *The carbon footprint of Indonesian palm oil production* (No. 25) (p. 10). Bogor, Indonesia.

Killeen, T. J., Harris, N. L., Brown, K., Netzer, M., & Gunarso, P. (2013). *Projections of Oil Palm Expansion in Indonesia, Malaysia and Papua New Guinea from 2010 to 2050*. *rspo.org* (pp. 89–112).

Koh, L. P., Miettinen, J., Liew, S. C., & Ghazoul, J. (2011). Remotely sensed evidence of tropical peatland conversion to oil palm. *Proceedings of the National Academy of Sciences of the United States of America*, *108*(12), 5127–32. doi:10.1073/pnas.1018776108

Langeveld, J. W. A., Dixon, J., van Keulen, H., & Quist-Wessel, P. M. F. (2013). *Analysing the effect of biofuel expansion on land use in major producing countries: evidence of increased multiple cropping*.

Limpert, E., Stahel, W. A., & Abbt, M. (2001). Log-normal Distributions across the Sciences: Keys and Clues. *BioScience*, *51*(5), 341–352.

Lugato, E., Panagos, P., Bampa, F., Jones, A., & Montanarella, L. (2013). A new baseline of organic carbon stock in European agricultural soils using a modelling approach. *Global Change Biology*, n/a–n/a. doi:10.1111/gcb.12292

Marwanto, S., & Agus, F. (2013). Is CO2 flux from oil palm plantations on peatland controlled by soil moisture and/or soil and air temperatures? *Mitigation and Adaptation Strategies for Global Change*, (12). doi:10.1007/s11027-013-9518-3

Melling, L., Goh, K. J., Chaddy, A., & Hatano, R. (2014). Soil CO2 Fluxes from Different Ages of Oil Palm in Tropical Peatland of Sarawak, Malaysia. In A. E. Hartemink & K. McSweeney (Eds.), *Soil Carbon* (1st ed., pp. 447 – 455). Cham: Springer International Publishing. doi:10.1007/978-3-319-04084-4

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

Melling, L., & Henson, I. (2011). Greenhouse gas exchange of tropical Peatlands–a review. *Journal of Oil Palm Research*, *23*(August), 1087–1095.

Miettinen, J., Hooijer, A., Shi, C., Tollenaar, D., Vernimmen, R., Liew, S. C., … Page, S. E. (2012). Extent of industrial plantations on Southeast Asian peatlands in 2010 with analysis of historical expansion and future projections. *GCB Bioenergy*, *4*(6), 908–918. doi:10.1111/j.1757-1707.2012.01172.x

Mutert, E., Fairhurst, T., & Uexkull, H. von. (1999). Agronomic management of oil palms on deep peat. *Better Crops International*, *13*(1), 22–27.

National Research Council. (2000). *Nutrient Requirement of Beef Cattle: Seventh Revised Edition: Update 2000*. Washington, D.C.

Noblet, J., Cozannet, P., & Skiba, F. (2012). Nutritional value and utilization of wheat dried distillers grain with solubles in pigs and poultry. In H. P.S.Makkar (Ed.), *Biofuel co-products as livestock feed - Opportunities and challenges*. Rome: FAO.

Omar, W., Abd Aziz, N., Tarmizi, A., Harun, M. H., & Kushairi, A. (2010). *Mapping of Oil Palm Cultivation on Peatland in Malaysia* (No. 529) (pp. 3–6). Kuala Lumpur, Malaysia.

Page, S. E., Morrison, R., Malins, C., Hooijer, A., Rieley, J. O., & Jauhiainen, J. (2011). *Review of peat surface greenhouse gas emissions from oil palm plantations in Southeast Asia* (No. 15). Washington D.C.

Ray, D. K., & Foley, J. A. (2013). Increasing global crop harvest frequency: recent trends and future directions. *Environmental Research Letters*, *8*(4), 44041. Retrieved from http://stacks.iop.org/1748-9326/8/i=4/a=044041

Ruesch, A. S., & Gibbs, H. K. (2008). *New IPCC Tier-1 Global Biomass Carbon Map For the Year 2000*. Oak Ridge, Tennessee. Retrieved from http://cdiac.ornl.gov/epubs/ndp/global_carbon/carbon_documentation.html

Scarlat, N., Martinov, M., & Dallemand, J.-F. (2010). Assessment of the availability of agricultural crop residues in the European Union: potential and limitations for bioenergy use. *Waste Management*, *30*, 1889–1897.

Schrier-Uijl, A., & Anshari, G. (2013). *Methods for determining greenhouse gas emissions and carbon stocks from oil palm plantations and their sourroundings in tropical peatlands*. *rspo.org* (pp. 169–191). Kuala Lumpur.

Schrier-Uijl, A., Silvius, M., & Parish, F. (2013). *Environmental and social impacts of oil palm cultivation on tropical peat–a scientific review* (pp. 131–168). Wageningen.

Siebert, S., Portmann, F. T., & Döll, P. (2010). Global Patterns of Cropland Use Intensity. *Remote Sensing*, *2*(7), 1625–1643. doi:10.3390/rs2071625

Spera, S. A., Cohn, A. S., VanWey, L. K., Mustard, J. F., Rudorff, B. F., Risso, J., & Adami, M. (2014). Recent cropping frequency, expansion, and abandonment in Mato Grosso, Brazil had selective land characteristics. *Environmental Research Letters*, *9*(6), 064010. doi:10.1088/1748-9326/9/6/064010

Uryu, Y., Mott, C., Foead, N., Budiman, A., Setiabudi, & Takakai, F. (2008). *Deforestation, forest degradation, biodiversity loss and CO2 emissions in Riau, Sumatra, Indonesia*. *One Indonesian Province's Forest and Peat Soil …*. Jakarta, Indonesia.

Verwer, C., Meer, P. Van Der, & Nabuurs, G. (2008). *Review of carbon flux estimates and other greenhouse gas emissions from oil palm cultivation on Tropical peatlands-Identifying the gaps in Knowledge*. Wageningen.

Wilhelm, W. W., Doran, J. W., & Power, J. F. (1986). Corn and Soybean Yield Response to Crop Residue Management Under No-Tillage Production Systems1. *Agronomy Journal*, *78*(1), 184. doi:10.2134/agronj1986.00021962007800010036x

Wösten, H., Ismail, A. B., & van Wijk, A. L. M. (1997). Peat subsidence and its practical implications: a case study in Malaysia. *Geoderma*, *78*(1-2), 25–36.