Carbon Emission Factors Subworkgroup

Low Carbon Fuel Standard (LCFS) Indirect Land Use Change Expert Workgroup

A Report to the California Air Resources Board

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Executive Summary

A key step of estimating the impacts of direct and indirect land use conversions as a result of increased biofuel demand is to estimate the amount and the duration of greenhouse gas (GHG) emissions as a result of land use and land use conversions in the various policy scenarios and the "base case" (i.e., no biofuel policy case). California Air Resources Board's (CARB) indirect land use change (ILUC) analysis for biofuels using the Global Trade Analysis Project (GTAP) model (CARB 2009)(CARB analysis hereafter) and a subsequent GTAP analysis by Tyner et al. (2010)(Tyner analysis hereafter) both characterize two kinds of emissions in their analyses: (1) carbon lost when forests or grasslands are cleared and converted into cropland, resulting in the loss of biomass and soil carbon stock; (2) foregone CO_2 sequestration by the forest converted to crop land.

The purpose of this report is to critically review the assumptions used in the CARB and Tyner analyses and to make recommendations for improvements in CARB's future analyses. We recognize that some of the recommendations may require more time and resources; therefore they are broken into "must", "short-term", and "long-term" recommendations.

This report summarizes our findings of the literature review and recommendation for the calculation of emission factors for the analysis of indirect land use change (ILUC). The list of topics our sub workgroup considers includes:

- Carbon stock values: including biomass carbon (C) with specific focus on forest biomass peatlands and soil C.
- Carbon emission rate upon land conversion: loss rates on conversion, harvested wood products, and non-Kyoto greenhouse gas (GHG) emissions.
- Emission factors of other indirect effects, including livestock emissions, rice cultivation, crop switching, and differences in on-farm energy and agrichemical use
- Uncertainty analysis

The above list only includes what the subgroup was able to examine given limited time and does not imply the exclusion of other important emission factors that we did not have the time or expertise to include in the report. For example, including fire emissions due to land clearing can significantly change the emission estimates in both the base case and biofuel scenarios. In addition, incorporating fertilizer use trend projections and emission factors associated with yield changes may also have big impacts on the emission estimates of base case and biofuel scenarios.

As expected, there are several cross-disciplinary issues across workgroups that need to be addressed in a consistent manner and require interactions with each other. Even though many of these interdisciplinary issues are relevant to the emission factors discussion, some of these issues are briefly summarized here and discussed more extensively in the other workgroups, especially the discussion of land cover types, emission time accounting, uncertainties, and emissions from co-products.

The Emission Factor subgroup conducted literature review and consulted with experts to reach the following recommendations:

	Must	Short term	Longer term
Biomass carbon stock	Use GIS to quantify forest carbon stocks for each GTAP region and AEZ combination based on the Winrock database and other recently published biomass maps	• Use GIS analysis to quantify cropland carbon stocks for each GTAP region and AEZ based on crop yield maps from Monfreda et al. (2008) • Conduct literature review of savanna, shrubland, and grassland biomass estimates to create an improved look-up table for the GTAP regions and AEZs	Refine estimates to account for different carbon stocks based on likelihood of conversion.
Soil carbon stock	• Use GIS to estimate soil C for region / AEZ using global datasets; supplementing with newer datasets such as STATSGO for U.S. and NSDB for Canada. • Use satellite-based land cover maps to produce soil carbon estimates for grassland, forest, cropland, or other land cover types that may be included in future CARB analysis.	Use GIS to overlay land cover and land management type, and produce soil carbon estimates by management type if data is available.	
Peatlands Emissions			The inclusion of peatland emission factors by land conversion type in CARB's future ILUC analyses.
Parameters for emission factor calculation		• Review assumptions of management practice effects on overall EF; particularly fire and "full tillage". • Incorporate new data on perennial storage in roots, soil and	

		aboveground biomass. Compile new data on disturbance effects and stages of sequestration at regional level.	
Long-term carbon storage in HWPs		Include HWP pool in the ILUC analysis to better reflect the timing of emissions. Confidence intervals should be included for all parameters in the HWP storage model to allow them to be included in an uncertainty analysis.	Consider consequential LCA by including the substitution by HWP for fossil energy and energy intensive materials as well as including energy used to produce wood products.
Other non-land conversion emissions		Consider a broader range of significant indirect emissions from land use changes such as, but not limited to, those related to livestock and rice production.	Consider the complexity of GHG emissions from crop switching
Non-Kyoto climate active GHG and aerosol emissions			Sensitivity analysis (short term) and uncertainty analysis (longer term) should be performed to explicitly consider the effects of non-Kyoto climate forcing gases and particles.
Uncertainty analysis		Report all stock and EF model parameters with uncertainty ranges or intervals. Use these distributions in GTAP's Systematic Sensitivity Analysis (SSA) feature to estimate the mean and standard deviation of the ILUC emission factor considering the combined uncertainty in economic and carbon accounting parameters.	Propagate uncertainty using Monte Carlo simulation.

1. Introduction

A key step of estimating the impacts of direct and indirect land use conversions as a result of increased biofuel demand is to estimate the amount and the duration of greenhouse gas (GHG) emissions as a result of land use and land use conversions in the various policy scenarios and the "base case" (i.e., no biofuel policy case). The underlying mechanisms of indirect land use change and resulting GHG emissions can be characterized in the figure below (Khanna et al 2010):

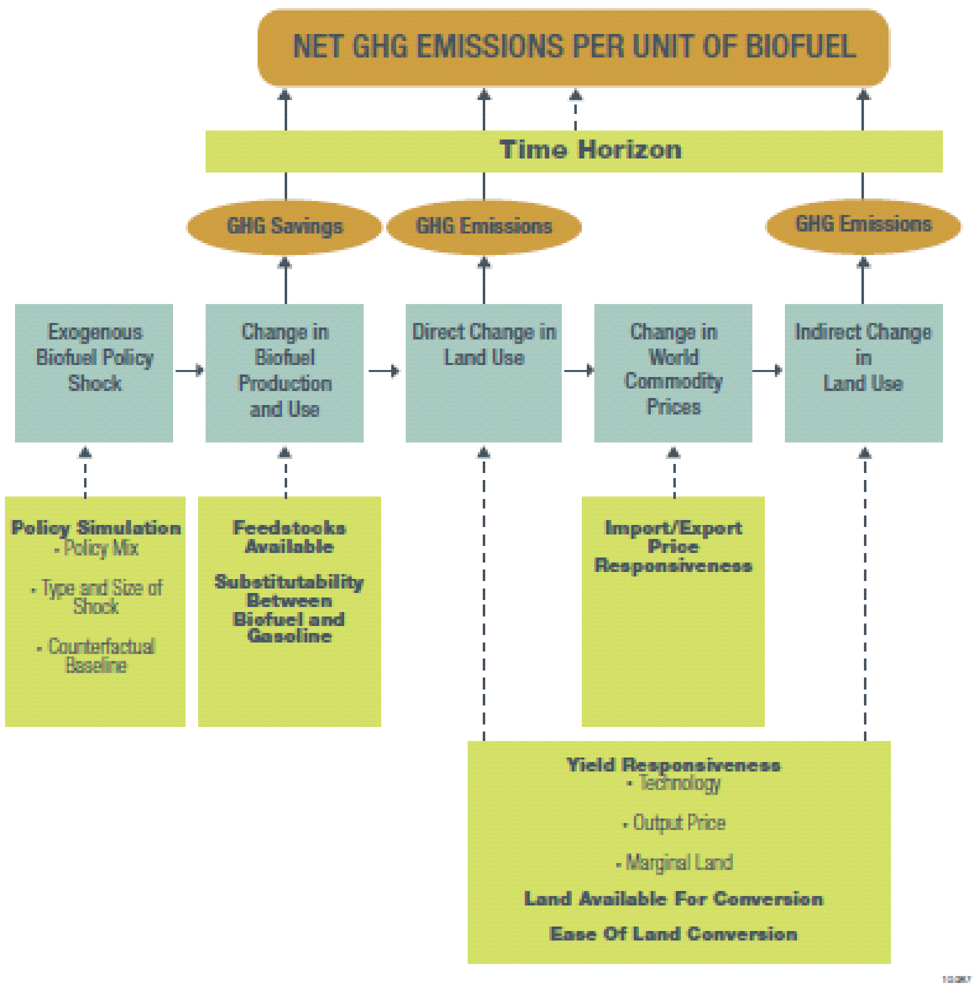

Figure 1.1: Modeling the land use change due to biofuels and their effect on GHG emissions. Source: Khanna et al. (2010).

California Air Resources Board's (CARB) indirect land use change (ILUC) analysis for biofuels using the Global Trade Analysis Project (GTAP) model (CARB 2009) and a subsequent GTAP analysis by Tyner et al. (2010) (hereafter CARB analysis and Tyner analysis, respectively) both characterize two kinds of emissions in their analyses: (1) carbon lost when forests or grasslands are cleared and converted into cropland, resulting in the loss of biomass and soil carbon stock; (2) foregone CO_2 sequestration by the loss of forest converted to crop land.

On the other hand, the US Environmental Protection Agency (US EPA) also conducted ILUC analysis in its analysis for the Renewable Fuel Standard (RFS2) and used a completely different set of models and assumptions for the calculation of ILUC emission factors (EPA 2010). It should be noted that there is a fundamental difference between the CARB vs EPA ILUC emission factors: CARB's ILUC emission factors are "add-on" values attached to the direct lifecycle emissions calculated from the GREET model for individual biofuel production pathways, whereas EPA's analysis use a consequential lifecycle analysis (LCA) approach that includes all major emission changes, including land-use change and non-LUC emissions, compared with a baseline scenario with no-RFS2 policy (Figure 1.2).

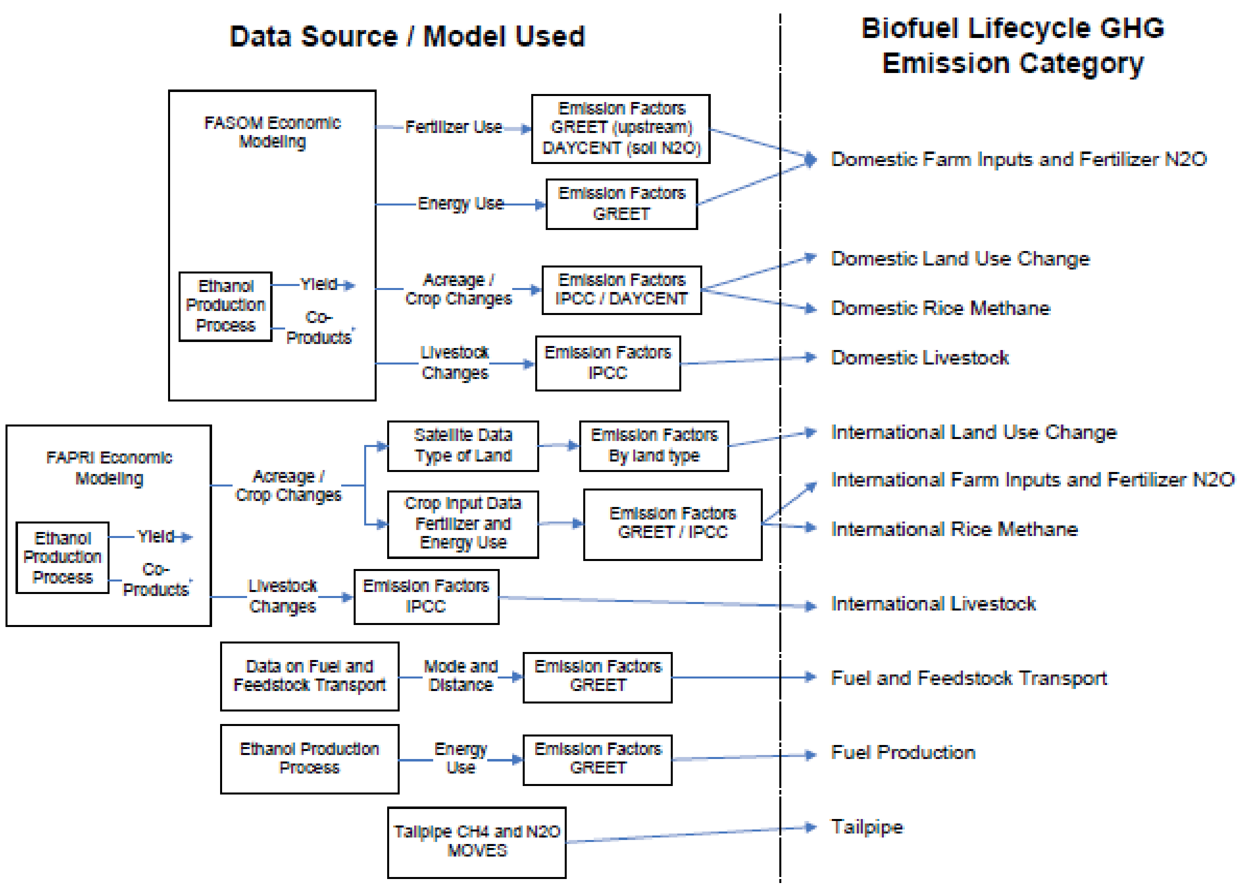

Figure 1.2: System boundaries and models used for EPA RFS2 analysis (Source: US EPA 2010).

1.1 Data and Assumptions of Emission Factors in the Previous CARB-GTAP Analyses

The data for biomass carbon and soil carbon stock values come from the Woods Hole Research Center (WHRC hereafter) data set, which is divided into ten world regions and thirty one ecosystem types. For any single region, the number of ecosystem type ranges from three to seven.

The conversion of land in each world region is based on an assumed percentage breakdown of ecosystems that are available for conversion. These percentage breakdowns are constant regardless of biofuel use scenarios. For example,

- Europe: temperate evergreen forest (25%), temperate deciduous forest (25%), boreal forest (25%), and temperate grassland (25%)
- United States: broadleaf forest (2%), mixed forest (34%), coniferous Pacific (2%), and grassland (62%)
- Latin America: tropical evergreen forest (3%), tropical seasonal forest (22%), tropical open forest (47%), temperate evergreen forest (3%), temperate seasonal forest (1%), grassland (24%), and desert (1%).

The resulting weighted average soil carbon stock concentration per hectare of land converted in each world region is illustrated in Figure 1.3 below. A similar graph can also be drawn for biomass carbon stock.

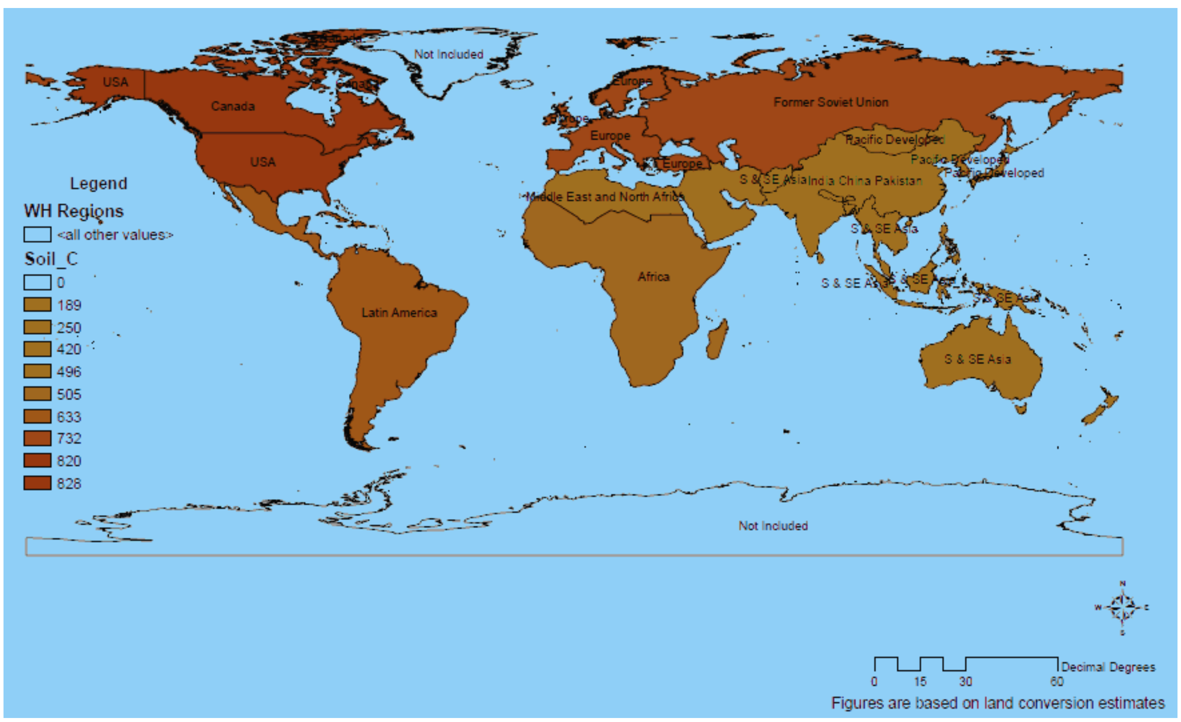

Figure 1.3: WHRC soil carbon stock (metric ton CO_2 per hectare) of available land for conversion in each of the ten world regions.

The CARB analysis further assumes that land conversion will result in the release of 25% of the soil carbon and 100% of biomass carbon at the time of land conversion (75% of forest biomass and 100% of grassland vegetation in Tyner analysis).

The GTAP model on the other hand is divided into nineteen world regions and Agro Ecological Zones (AEZs). Each AEZ shares common climate, precipitation and moisture conditions (Figure 1.4).

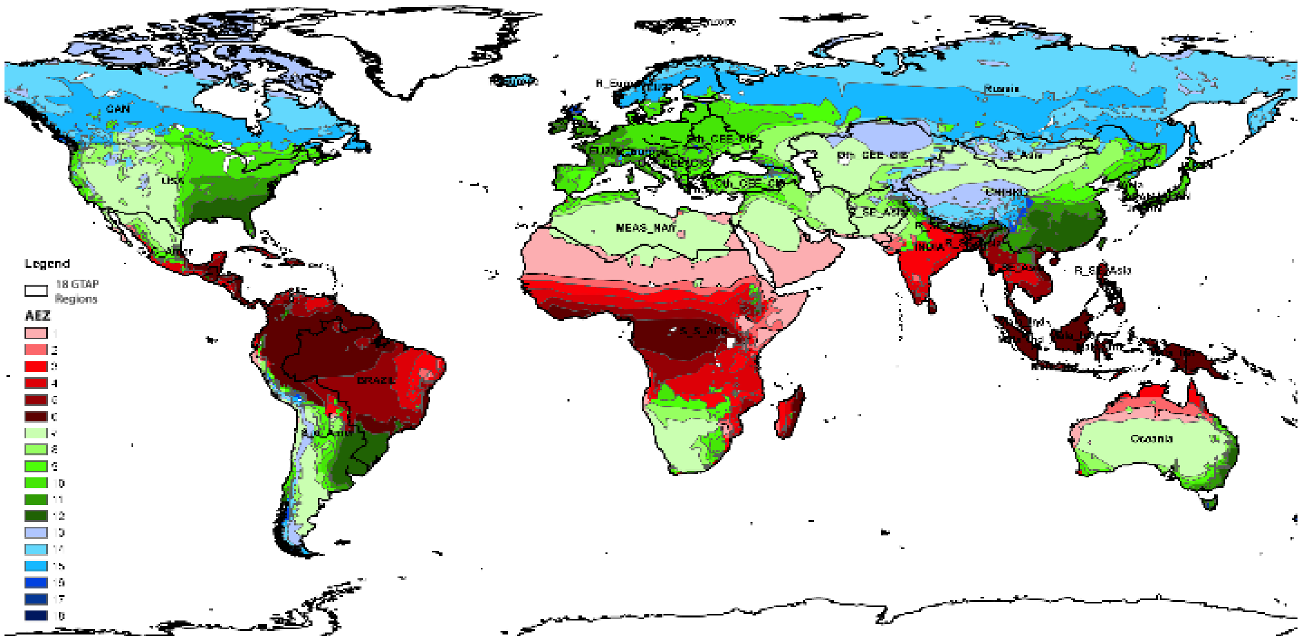

Figure 1.4: GTAP's nineteen world regions and Agro Ecological Zones (AEZs).

2. Biomass Carbon Stock

CARB estimates biomass and soil carbon stocks using the WHRC database, which is based on an extensive literature review by R.A. Houghton (See Gibbs et al. 2007 for synthesis of data sources). The WHRC data is not spatially explicit but rather provides a look-up table for broad regions that is applied to all of the AEZs in a region.

Table 2.1: Regional comparison of number of regions in the Winrock and WHRC databases with those in GTAP.

GTAP	Winrock*	WHRC	Winrock*
United States	49	United States	49
Canada	13	Canada	13
Sub Saharan Africa	85	Africa	85
EU 27	26	Europe	43
E Europe and Rest of Former Soviet Union	10		
Rest of European Countries	10		
Russia	88	Former Soviet Union	93
Brazil	29	Latin America	124
Central and Caribbean Americas	39		
S & Other Americas	56		
Mid East & N Africa	45	N Africa	45
E Asia	4	Pac Developed	15
Oceania	10		
Japan	1		
China & Hong Kong	31	China/India/Pakistan	67
India	35		
Rest of SE Asia	172	S & SE Asia	222
Rest of S Asia	6		
Malaysia & Indonesia	45		

2.1 Winrock Biomass Carbon Database

2.1.1 Winrock Forest Biomass Carbon Database

For its indirect effects analysis for RFS2, the EPA relies on a model created by Winrock International (Harris et al 2008). Winrock synthesized a range of forest biomass carbon datasets, each created using different methodologies. The global vegetation carbon map created by Ruesch and Gibbs (2008) is used to fill gaps where more detailed datasets are unavailable. All datasets are published except for a set of satellite-based data developed by Sassan Saatchi and Winrock that is still being finalized.

Most of the datasets are spatially-explicit, which is a major advantage over the WHRC look-up table because it provides estimates tailored as much as possible to the regions of interest. The spatially-explicit data also have advantages over the estimates compiled by Sohngen (Section 2.2) because in most cases they are based on more data points and better account for the variation of carbon stocks across the landscape. For the EPA analysis, Winrock used a geographic information system (GIS) to clip the carbon maps to each country / administrative unit and calculate weighted average carbon stocks. **We recommend using the Winrock database (with minor refinements to account for recently published carbon maps) to estimate forest carbon stocks for each GTAP AEZ and region.**

Note that it is important to capture the carbon stocks of the forests most likely to be cleared rather than an average for the entire region as is currently used by CARB (based on WHRC). Please see the report by the Land Cover Types Subgroup for recommendations on methods to exclude forests that are inaccessible and develop conversion probabilities.

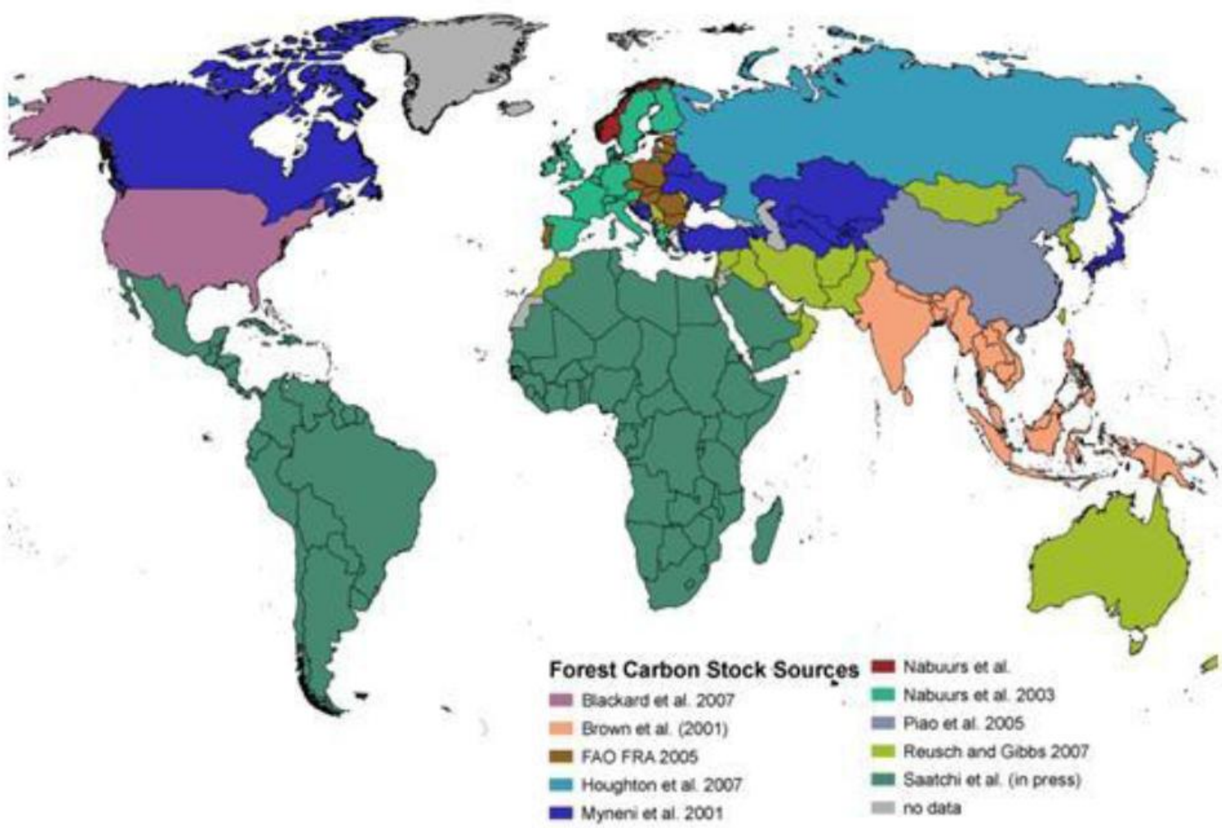

Figure 2.1: Range of data sources used by Winrock (Harris et al. 2008) to estimate forest carbon stocks.

2.1.2 Winrock Cropland Biomass Carbon Database

Winrock uses a single IPCC default value for all annual croplands and all plantations. While cropland carbon stocks do not vary as much as forest carbon stocks, they do vary according to yields and crop type. **We recommend using crop yield maps by Monfreda et al (2008) to provide regionally-specific biomass estimates for different crop types.** Crop biomass values could be scaled according to yield as in Gibbs et al. (2008) or by assuming that calculations of net primary productivity (NPP) based on yield data are equivalent to the standing carbon stock as in West et al. (2010).

2.1.3 Winrock Grassland, Savana and Shrubland Biomass Estimates

Winrock used the IPCC default values for grassland, savannas and shrubland for all countries except Brazil where they followed de Castro and Kauffman (1998). Similar to croplands, biomass values vary across landscapes for these cover types but the variation is less than with forests. **We recommend reviewing the literature for more recent efforts to estimate and map these carbon stocks in different regions, to help provide more detailed values for key regions and AEZs.**

2.2 Alternative Approach to Forest Biomass

Data compiled by Prof. Brent Sohngen at OSU can be used to derive an alternative set of emissions factors to the currently used Woods Hole dataset. [1] OSU, as part of the Global Timber Market and Forestry Data Project produced country-specific datasets listing accessible and inaccessible forest areas with the respective accessible and inaccessible above ground tons carbon. OSU datasets are available for 150 countries across all GTAP regions. Furthermore, the datasets cover 18 AEZ regions. All land area and carbon data sets are available online.[2] The individual forest inventory source data varies by country. FAO data was used for most countries. For the US inventory data was taken from the USDA FIA, and for the remaining countries including China, inventory data was sourced from official country-specific publications.

According to Brent Sohngen, accessible and inaccessible lands are delineated differently depending on the geographic region. For the US, accessibility is a function of timber demand and price as outlined in Sohngen and Sedjo (1999).[3] For Europe, all forests are deemed accessible and for the tropics accessibility is based on proximity of forestland to roadways.

Table 2.2 below illustrates the available data with the US as an example and compares the derived emissions factor to the Woods Hole data. For the US, the data would suggest forest

[1] Global Timber Market and Forestry Data Project, Version 5, 2007, Sohngen and Tennity, OSU. Data has been compiled and made available with the financial assistance of the US EPA, Climate Analysis Branch.

[2] http://aede.osu.edu/people/sohngen.1/forests/GTM/data2.htm

[3] Sohngen, Brent and Roger Sedjo, "Potential Carbon Flux from Timber Harvests and Management in the Context of Global Timber Market"; Report funded by the US Department of Energy and Resources for the Future, 1999, available at http://www-agecon.ag.ohio-state.edu/people/sohngen.1/forests/c_stor.pdf

emissions factors for accessible land of 48 t C/ha, 96 t C/ha for inaccessible land, and a combined factor of 62 t C/ha, compared to the Woods Hole factor of 113 t C/ha. As part of this analysis we extracted data for several other key countries and compared the derived emissions factors to the Woods Hole factors. The results are listed in Table 2.3. Compared to the Woods Hole factors, the combined (accessible land and inaccessible land) factors are the same for Brazil and Japan but, similar to the factor for the US, lower for Canada, India, and Russia.

In summary, the OSU emissions factors provide a higher resolution than the Woods Hole factors since data is available for all GTAP regions and by AEZ. However, some of the inventory data sets date back to 1990 and the accessible/inaccessible land delineation for the tropics is likely too simplified for CARB's purposes. The OSU factors are comparable to or lower than those from Woods Hole for above ground forest biomass. However, more updated datasets should be analyzed to confirm this finding.

Table 2.2: OSU Data for the United States

United States AEZ	Accessible ha	In-accessible ha	Total (ha)	Accessible Million t C	In-accessible Million t C	Total Million t C	Accessible t C per ha	In-accessible t C per ha	Total t C/ha	Woods Hole - GTAP 2010 t C/ha
1	0	0	0	0	0	0	0.0	0.0	0.0	
2	0	0	0	0	0	0	0.0	0.0	0.0	
3	0	0	0	0	0	0	0.0	0.0	0.0	
4	0	0	0	0	0	0	0.0	0.0	0.0	
5	0	0	0	0	0	0	0.0	0.0	0.0	
6	0	0	0	0	0	0	0.0	0.0	0.0	
7	1,262,860	1,541,948	2,804,808	49	113	162	39.1	73.3	57.9	
8	5,051,441	6,167,791	11,219,232	198	452	650	39.1	73.3	57.9	
9	3,367,627	4,111,861	7,479,488	132	301	433	39.1	73.3	57.9	
10	36,344,528	12,030,874	48,375,401	1,862	994	2,856	51.2	82.6	59.0	
11	21,507,454	7,711,778	29,219,231	1,223	697	1,920	56.9	90.4	65.7	
12	68,734,685	21,424,930	90,159,614	3,138	2,662	5,800	45.7	124.3	64.3	
13	1,894,290	2,312,922	4,207,212	74	170	244	39.1	73.3	57.9	
14	1,894,290	2,312,922	4,207,212	74	170	244	39.1	73.3	57.9	
15	420,953	513,983	934,936	16	38	54	39.1	73.3	57.9	
16	1,473,337	1,798,939	3,272,276	58	132	189	39.1	73.3	57.9	
17	0	0	0	0	0	0	0.0	0.0	0.0	
18	0	0	0	0	0	0	0.0	0.0	0.0	
							48	96	62	113

Table 2.3: Carbon Stock Comparison

	Accessible Land (t C/ha)	Inaccessible Land (t C/ha)	Accessible and Inaccessible Land (t C/ha)	Woods Hole (t C/ha)
US	48	96	62	113
Brazil	102	103	102	102
Canada	30	30	30	74
India	29	70	60	139
Russia	17	46	39	65
Japan	46	108	74	75

2.3 Recommendations

Must do

- Use GIS to quantify forest carbon stocks for each GTAP region and AEZ combination based on the Winrock database and other recently published biomass maps

Short term

- Use GIS analysis to quantify cropland carbon stocks for each GTAP region and AEZ based on crop yield maps from Monfreda et al. (2008)
- Conduct literature review of savanna, shrubland, and grassland biomass estimates to create an improved look-up table for the GTAP regions and AEZs

Longer term

- Refine estimates to account for different carbon stocks based on likelihood of conversion.

3. Soil Carbon Stock

CARB used the WHRC database to characterize soil carbon stock values. As shown in Figure 1.1, the spatial resolution of such data is rather coarse compared with the spatial resolution of the projected land use change from the GTAP model (Figure 1.2).

3.1 Soil C Databases

Winrock used data from the Harmonized World Soil Database (HWSD) in its analysis for the RFS2. This database combines four databases to create an international soil map (Soil Map of the World, SOTER Regional Studies, European Soil Database, Soil Map of China 1:1 Million Scale). Figure 3.1 below shows the coverage of each of the datasets. Further expansion and update of the HWSD is foreseen for the near future, notably with the excellent databases held in the USA: Natural Resources Conservation Service US General Soil Map (STATSGO) http://www.ncgc.nrcs.usda.gov/products/datasets/statsgo, Canada: Agriculture and Agri- Food Canada: The National Soil Database (NSDB) http://sis.agr.gc.ca/cansis/nsdb and Australia: CSIRO, ACLEP, Natural Heritage Trust and National Land and Water Resources Audit: ASRIS http://www.asris.csiro.au/index_other.html, and with the recently released SOTER database for cntral Africa (FAO/ISRIC/University Gent, 2007).

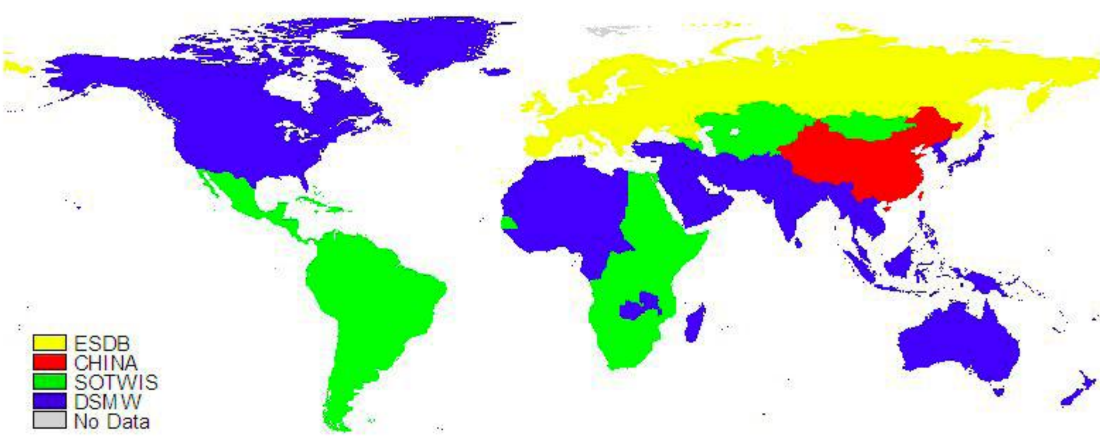

Figure 3.1: Sources underlying the Harmonized World Soil Database. ESDB: the European Soil Database; CHINA: Soil map of China 1:1 Million Scale; various regional SOTER databases (SOTWIS Database), and the Soil Map of the World.

The HWSD provides information for 15,773 soil mapping units at 30 arc second resolution. The database shows the composition of each soil mapping unit and standardized soil parameters for top (0-30 cm) and subsoil (30-100 cm). The database does not provide the total carbon per ha, but it does provide information to calculate the total C /ha using the following equation from Guo and Gifford (2002):

$$C_t = BD \times C_c\% \times D$$

Where,

C_t = total soil carbon concentration (t/ha)
BD = soil bulk density (g cm^{-3})
C_c% = soil concentration (%)
D = soil sampling depth (cm)

Some areas lacked information on bulk density; for these areas the following equation was used:

$$BD = \frac{100}{\frac{\%OM}{0.244} + \frac{100-\%OM}{1.64}}$$

Where,

%OM = organic matter (or loss by ignition) as a percentage of soil dry mass.

The map below (Figure 3.2) shows the carbon estimates based on the HWSD 0-100 cm data. The HWSD collected information about the topsoil (0-30cm) and subsoil (30-100cm).

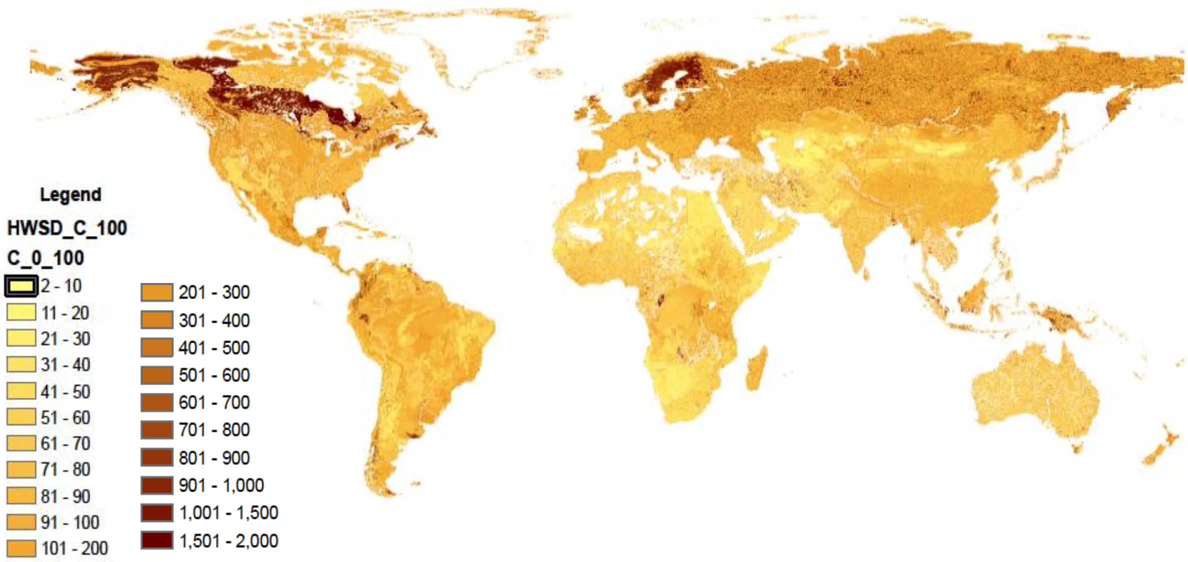

Figure 3.2: Soil carbon (t C/ha) calculated for 0-100 cm depth based on data from HWSD.

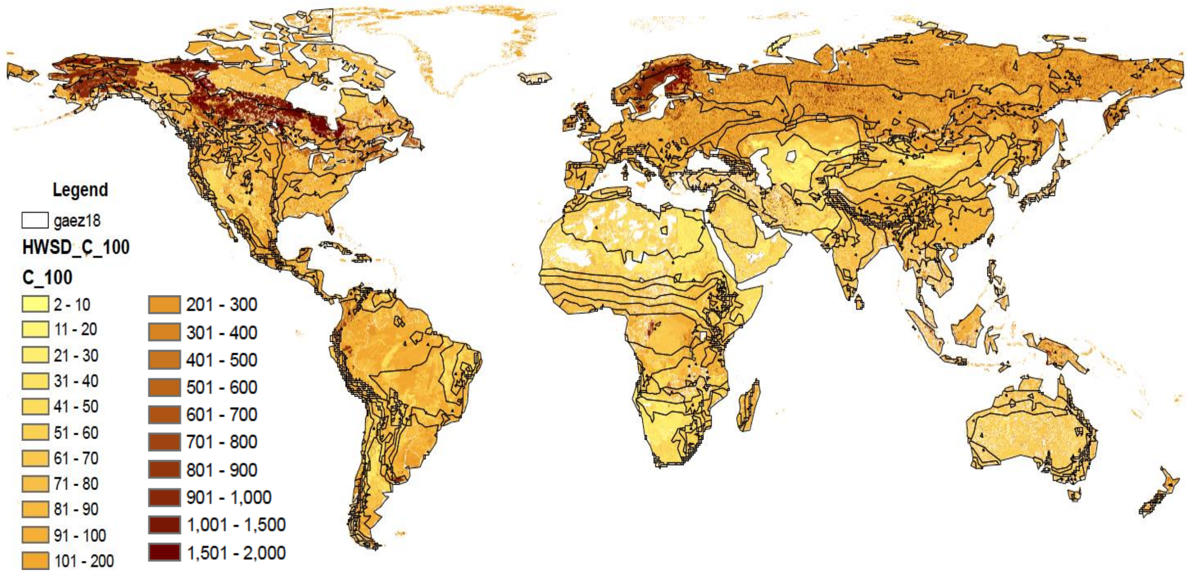

Figure 3.3: Soil carbon (t C ha-1) 0-100 cm depth by AEZ

3.2 Land Conversion

The effect of land management and land cover type has considerable effect on soil carbon and nitrogen. In a meta analysis done by Guo and Gifford, the impacts of land management on soil carbon and nitrogen was reviewed from 74 different publications. The study concluded that certain land conversion (such as native forest to crop or pasture to crop) will release a great deal of carbon, but other land conversions (such as crop to secondary forest) can have a positive impact on soil carbon stock (Guo Gifford 2002). In a separate study, the land conversion from forest to cultivated land shows the average loss to be 22% - with the most soil carbon loss in the first 20 years (Murty et al 2002). Below lists the impacts of land conversion on soil carbon (Guo Gifford 2002):

pasture → plantation: -10%
native forest → plantation: -13%
native forest → crop: -42%
pasture → crop: -59%
native forest →pasture: +8%
crop → pasture: +19%
crop → plantation: +18%
crop → secondary forest: +53%
native forest or pasture → broad leaf plantation: ~0
native forest or pasture → pine plantation: -12-15%

The above list exemplifies the importance in considering the difference in carbon losses or gains of different conversion types (though the above example does not differentiate between climatic regions), as opposed to a uniform loss rate of 25% assumed in the CARB analysis. A comprehensive review of land conversion emission factors is discussed in Section 5.

3.3 Recommendations

Most do

- Use GIS to estimate soil C for region / AEZ using global datasets; supplementing with newer datasets such as STATSGO for U.S. and NSDB for Canada.
- Use satellite-based land cover maps to produce soil carbon estimates for grassland, forest, cropland, or other land cover types that may be included in future CARB analysis.

Short term

- Use GIS to overlay land cover and land management type, and produce soil carbon estimates by management type if data is available.

4. Peatlands Emissions

The CARB and Tyner analyses do not explicitly represent peatlands as one of the land cover types, nor do they consider the conversion of peatlands for agricultural and other purposes. In the Winrock analysis for RFS2 (Harris et al. 2009), peatland areas cover 2-44% and 2-22% in some of the corresponding administrative regions in Indonesia and Malaysia, respectively. The emission rates are assumed to be 20 t C/ha/yr using 80 cm drainage depth. The emission factors are calculated for 30 and 80 years and the cumulative emissions of peatland conversion to croplands are 600 and 1600 t C/ha, respectively.

In an earlier study published in Science, Fargione et al. (Fargione et al. 2008) estimated the CO_2 released from drained peat soils in tropical rainforest Southeast Asia over 50 years is 941 t C/ha (750, 145, 797, and 47 in soil, aboveground, belowground, and root respectively), equivalent to 18.8 t C/ha/yr. Fargione et al. (2008) acknowledge that this underestimates the CO_2 that would be released if drainage were to be sustained for longer than 50 years.

Both studies regard the magnitude of peatland emissions as highly uncertain.

4.1 Peatland Areas and Carbon Stocks

There are several estimates of the total area of peatland as well as the overall carbon stock and emissions. The values from various journals and articles show a range of 3-10% of global terrestrial area coverage is peatland (Jaenicke et al 2008, Hadi et al 2001). Studies show that although peatland does not occupy significant land area, it contains considerable carbon and nitrogen (15-33% of the global terrestrial soil carbon)(Page et al 2008, Furukawa et al 2000). Although temperate and boreal regions have the greatest extent of peatland, Southeast Asia has the greatest extent within tropical peatlands: with 56% of tropical peatland in Southeast Asia (Page et al 2008). Table 4.1 below shows a breakdown of peatland carbon pools and carbon stock density per unit area of land.

Table 4.1: Estimates of global and tropical peatland carbon pools. Source: IPCC WG1, Vitt et al (2000) and Page et al (2010).

	Minimum	Best estimate	Maximum	Carbon density (t C /ha)
Tropical forest (Gt)		216		122
Global peat carbon pool (Gt)	598	610	618	
Boreal/temperate peat carbon pool (Gt)	517	521.4	526	900-1390
Tropical peat carbon pool (Gt)	82	88.6	92	1400-2110
Southeast Asian peat carbon pool (Gt)	66	69	70	2200-3500

4.2 Conversion and Emissions

Peatland conversion has increased significantly in the past few decades, in Southeast Asia alone 47% of peatland (12.9 Mha) was deforested by 2006, with an estimated deforestation rate in Indonesia of 1.3% per year (Hooijer et al 2006). Within that 47%, the majority (67%) was deforested for small-scale agriculture, and the remainder was split between large-scale agriculture or cleared and burnt (Hooijer et al 2006). From 1997-2000 the area logged in Indonesia increased by 44% (Page et al 2001). The estimate of CO_2 emissions from decomposition of drained peatlands is between 355 Mt/yr ~ 855 Mt/yr (Hooijer et al 2006). Hooijer's emissions estimate comes from peatland extent, thickness, projected and current land use, water management practices and decomposition rates. Indonesia is responsible for 82% (7954 Mt) of the Southeast Asia CO_2 emissions in 2006 (from ongoing peat decomposition) (Hoojier et al 2008).

Hoojier et al (2006) found that unit CO_2 emission is a linear function of groundwater depth and % area drained in converted land. Peatland emissions can be estimated based on the following equation:

$$CO2\ emission\ (t\ CO_2/y) = LU\ Area \times D\_Area \times D\_Depth \times CO2\_1m$$
(Equation 4.1)

Where,

LU Area = peatland area with specific land use [ha]
D_Area = drained area within peatland area with specific land use [fraction]
D_Depth = average groundwater depth in drained peatland area with specific land use [m]
CO_2_1m = CO_2 emissions occurring at an average groundwater depth of 1m = 91 [t CO_2 /ha/yr]

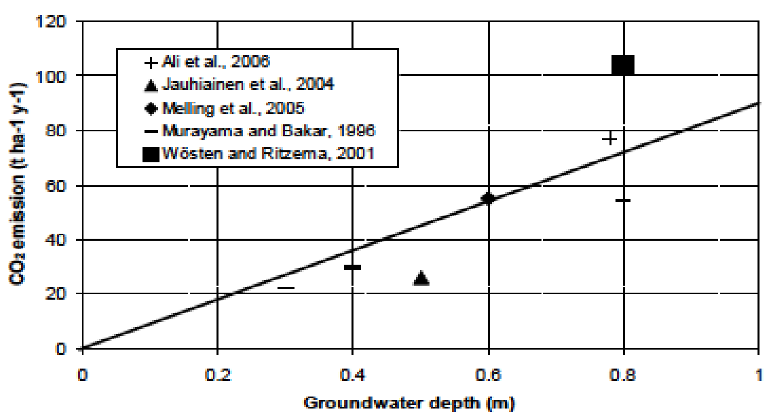

Figure 4.1: Linear relation between groundwater depth in peatland and CO_2 emission caused by peat decomposition. The line has been fitted through published measurements in agricultural areas in peatland, including oil palm plantations. Source: Hoojier et al (2006)

Peatland drained to 0.95 m as typically needed for croplands, including palm plantations, on average emits 86 t CO_2 /ha/yr. Cropland and palm plantation keeps average water tables always below 0.7 m, but they are often as deep as 1.2 m on average. For small-scale agriculture such as mixed cropland and shrubland, peatland is typically drained to 0.6m depth for 88% of the area,emitting 48 t CO_2/ha/yr. For shrubland in recently cleared and burnt area, the drainage depth is typically 0.33 m over half of area and emits 15 t CO_2/ha/yr (Table 4.2).

Table 4.2: Typical tropical peatland emissions (tonnes CO_2/ha/yr). Source: Hooijer et al (2010)

Large croplands, including plantations	86 (73-100)
Mixed cropland / shrubland: small-scale agriculture	48 (27-73)
Shrubland; recently cleared & burnt areas	15 (6-27)

4.3 Recommendation

Longer term

Due to the high likelihood of peatland conversion for agricultural purposes in some regions (Hooijer et al 2010), and the high carbon stocks and emission rates associated with peatland conversion, we recommend the inclusion of peatland emission factors by land conversion type in CARB's future ILUC analyses. However, such implementation requires a better understanding of *when* and *how* much peatland area will be converted under various scenarios, and this capability may not be possible under CARB's current ILUC model structure in the short term. This topic is discussed in greater detail in a separate report by the Land Conversion Type sub-workgroup.

5. Parameters for Emission Factor Calculation

Emission factors for each land cover conversion scenario have been evaluated for ILUC studies conducted by Winrock for EPA Renewable Fuel Standard (RFS2) and CARB for the Low Carbon Fuel Standard (LCFS). We also reviewed additional approaches to calculating EF by other methods[4]. EPA and CARB presented total emission factors for conversion and reversion emissions per land type considered for a 30 year timeframe. However, the carbon data, models and resulting emission factors are not directly comparable. Therefore we focus on the assumptions involved in the emission factor calculations. Treatment of uncertainty in emission factors is discussed in Section 9.

5.1 Comparisons of Assumptions for Emission Factors

IPCC Tier 1 through 2 and stock change emission factor methods are used in both the Winrock RFS2 analysis and the CARB LCFS ILUC calculations. Winrock supplemented this method with the loss/gain method for forest reversion estimates. EFs are calculated per pool and are variable. Generally, default stock change factors were applied in both assessments. IPCC recommends reporting carbon stock changes in five pools: aboveground biomass, belowground biomass, dead wood, litter, and soil organic carbon. However, data are not available for all land types; we assume no change in pools for which data is unavailable. Also, a decrease in one pool may be offset by increases in another pool, e.g., biomass pools decline after a disturbance such as fire but litter and dead wood pools can increase. Therefore, the change in a single pool can be greater than the net change in the sum of the pools. Table 5.1 identifies each pool or factor, the general IPCC GHG conversion response, and calculation steps used by Winrock and CARB.

Table 5.1: Carbon pools for emission factor calculations

C Pools	LUC Assumption and IPCC Default (IPCC LULUCF, 2006) [5]	EPA vs. CARB Treatment for Emission Factor	Suggested Approach for CARB
Aboveground Biomass (AG)	Emitted by conversion, forgone by harvested wood or sequestered through new growth. Default oxidation 100% grassland; 75% forests; 20 years reversion for grassland/pasture. Default method: stockchange approach AG (t C per ha); dry weight converted to CO_2.	CARB uses stock change method; EPA uses combination stock change/ and loss/gain method. Pool estimated by EPA is a combination of carbon maps/ground measurements; CARB used WHRC historical averages. Both included forgone sequestration and reversion. Forests considered gaining were included in both CARB and EPA EF.	Match EF to AEZ using suggested approach in Chapter 3.

[4] Approaches in O'Hare et al., (2009), Golub et al., (2010), Hertel (2010), and others on time treatment and uncertainty is covered by other EWG sub-groups

[5] IPCC (2006) Best Practice guide for C stock estimates and emissions is somewhat different to the AFLOU method used in the CDM and REDD accounting for national inventory reporting.

C Pools	LUC Assumption and IPCC Default (IPCC LULUCF, 2006) [5]	EPA vs. CARB Treatment for Emission Factor	Suggested Approach for CARB
Belowground Biomass (BG) including roots	Carbon in all biomass of live roots. Fine roots of less than 2 mm diameter are excluded, because these often cannot be distinguished empirically from soil organic matter or litter. Can include below ground part of stump. Emitted by conversion, foregone by reduced tillage or sequestered by using different crop practices	CARB did not account for roots. EPA included BG biomass separated by roots and SOC.	CARB to consider Winrock inclusion roots in BG estimate.
C in Litter	Emitted by natural decay; sped up by conversion by fire and in some cases re-incorporated back to the soil by sequestration	Not accounted for by EPA or CARB	Litter (t C/ha) Carbon in all non-living biomass with a diameter less than the minimum diameter for dead wood (e.g. 10 cm), lying dead in various states of decomposition above the mineral or organic soil.
Dead Wood (DW)	Dead wood includes wood lying on the surface, dead roots, and stumps larger than or equal to 10 cm in diameter or any other diameter used by the country[6]. Carbon in all non-living woody biomass not contained in the litter, either standing, lying on the ground, or in the soil. Emitted by natural decay; converted to biochar for sequestration back into SOC or burned to increase initial emissions.	Not accounted for by EPA or CARB	DW (usually omitted except for forest systems can be substantial; t C/ha).
Soil Carbon (SOC)	Assumed linked to conversion response of aboveground biomass. SOC (as 25% of AG estimate in t C/ha or using regional data as t C/ha). IPCC default 30 cm depth; 20 years to reach equilibrium. Organic carbon in mineral and organic soils (including peat) to a specified depth chosen by the country and applied consistently through the time series. Fine roots < 2mm can be included. Inorganic carbon is not included.	BG =SOC measurements in top 30cm of soil followed by EPA and use of HWSD; CARB estimates include top 100cm (non –IPCC). Both used IPCC default for 20 years for forests reach equilibrium.	Match HWSD to AEZ as recommended in Chapter 3. Consider long term storage of roots in grasslands.

[6] Often ignored, or assumed in equilibrium, this carbon pool can contain 10-20% of that in the AGB pool in mature forest (Delaney et al., 1998). However, in immature forests and plantations both standing and fallen dead wood are likely to be insignificant in the first 30-60 years of establishment (Watson, UNDP, 2009).

5.2 IPCC Emission Factor Method

IPCC provides a decision tree method to estimate emission factors from land conversion in the Agriculture Forest and Other Land Use (AFOLU) and additional guidelines in the IPCC LULUCF Good Practice Guide (GPG) to incorporate default data and methods (IPCC GPG, 2000, IPCC AFOLU Chapter 4, 2006). The IPCC Tier 1 emission factor method assumes that the net change in the carbon stock for litter (forest floor), dead wood and soil organic carbon (SOC) pools is zero, but using national accounting (Tier 2 and above) the belowground biomass, litter, dead wood and SOC should all be counted unless the country chooses not to count a pool that can be shown not to be a source. Therefore Tier 1 can only be applied if the litter, dead wood and SOC pools can be shown not to be a source using the methods outlined by IPCC. Tier 1 can also only be applied if forest management is not considered a key category, which can only be the case with "forests remaining forests".

Tier 1 sets carbon stock change to zero if the average age of the tree population is 20 years or less; otherwise carbon stock change in biomass growth is assumed equal to loss. Also, estimates of aboveground biomass stock change for grasslands are meant to be used only to calculate emission factors from burning.

Stock Change Approach

$$\mathbf{EF = LUC_{biomass} + Change\ C_{soil,\ LUC} + Change\ C_{lost\ seq\_luc} + ELUC_{fire} + ELUC_{rice}}$$

where:

EF = emission factor for converting one hectare of land to land use LUC; t CO_2 per ha

$LUC_{biomass}$ = change in biomass carbon stocks as a result of land use change LUC; t CO_2 per ha

Change $C_{soil,\ LUC}$ = change in soil carbon stocks (top 30 cm) as a result of land use change LUC; t CO_2 per ha

Change $C_{lost\ seq\_luc}$ = lost forest sequestration resulting from land use change (if applicable); t CO_2 per ha

$ELUC_{fire}$, = non-CO_2 GHG emission associated with land clearing with fire (if applicable); t CO_2 per ha

$ELUC_{rice}$ = non-CO_2 GHG (CH_4) emission associated with rice cultivation (if applicable); t CO_2 per ha

Both CARB and EPA modified this approach somewhat. For example, Winrock included forgone sequestration rates from long term monitoring plots from temperate and tropical forests. CARB estimates foregone C is equivalent to lost annual sequestration (Mg C/ha/y) in existing forests with the exception in EU and FSU, assumed to be regrowing. Both CARB and EPA did not include rice emissions in the LUC model, although EPA included rice methane emissions elsewhere in the LCA (Harris et al., 2009), see Chapter 7 for more discussion.

Estimates of emissions/removals using model-based approaches derived from the interaction of multiple equations that estimate the net change of biomass stocks within the models. In many instances, the carbon stocks in forests may change without a change in forest area and therefore gaining carbon. Examples of losses include of biomass associated with selective wood harvest, forest fragmentation, ground fires, shifting cultivation, browsing, and grazing (Barlow et al., 2003, Houghton, 2005).

5.3 Stock Change Factors and improved practice

Land use practice is critical to include, and particular assumptions, e.g., fire clearance, full tillage practice, can increase emission factors. The four main factors in IPCC listed in Table 5.2 were included by EPA and CARB in each emission factor differently. Although IPCC notes high uncertainty associated with default stock change values, Tier 1 estimates do not include uncertainty for each model parameter.

Table 5.2: Assumptions of stock change factors

Stock Change Factors	LUC Assumption and/or IPCC Default	EPA and CARB EF	Suggested Improvement
Harvested Wood Product (HWP)	Harvested Wood Products (HWP) for export counted as stored carbon as a percentage for ILUC emission factors.	CARB assumes 0% of carbon stored in HWP, Tyner analysis assumes 25% of forest biomass carbon stored in HWP within 30 years of modeling period. EPA evaluated HWP but concluded the numbers are so small that it was not included in EF.	Emitted differently depending on the product so counted as a negative emission for a set period of time. See Chapter 6.
Fire	Considered a conversion response as an immediate emission.	EPA employed IPCC default factors for N_20 and CH_4 combustion. Not included in CARB analysis.	Consider yearly burning practice and improved practice (i.e.CARB credit for Brazil sugarcane) in direct LCA. Include combustion factors per Winrock.
Disturbance	Considered as a conversion response as an immediate emission. LU disturbances by insect, disease, other abiotic factors, e.g. drought, storm, insect disturbance, etc.vary and usually not included in calculation.	EPA and CARB did not include ‘other’ disturbance	
Forest Management; LU management	Considered in a variety of response variables for immediate emission or sequestration value for reduced or ‘better’ management practice as opposed to conventional methods	EPA assumed full tillage and medium inputs for cropland management.	Consider IPCC default values matched to new land types in GTAP

IPCC uses a decision tree approach for any given activity in any year. Utilized for national reporting under the Marrakesh Accords, this method recognizes that land can have multiple activities and that allocation to shifts in land use can be problematic. Table 5.3 lists relative stock change factors for different management practice.

Table 5.3: IPCC Relative Stock Change Factors (FLU, FMG, AND FI) (Over 20 years) for Different Management Activities on Cropland

Level	Temperature regime	Moisture regime	IPCC defaults	Error	Description
Long term cultivated	Temperate/ Boreal	Dry	0.8	9% (+/-)	Represents area continuously managed for >20 yrs, to predominately annual crops. Input and tillage factors are also applied to est. C stock changes. LU factor est. relative to use of full tillage and nominal ('medium') carbon input levels.
		Moist	0.69	12% (+/-)	
	Tropical	Dry	0.58	61% (+/-)	
		Moist/Wet	0.48	46% (+/-)	
	Tropical montane	n/a	0.64	50% (+/-)	
Paddy Rice	All	Dry and Moist/Wet	1.1	50% (+/-)	Long term (>20 year) annual cropping of wetlands. Can include double- cropping with non-flooded crops. Tillage and input factors not used for paddy rice.
Perennial/Tree	All	Dry and Moist/Wet	1	50% (+/-)	Long term perennial tree crops such as fruit and nut trees, coffee and cacao.

IPCC notes uncertainty and variability in management factors for several reasons: i) the list of candidate activities is not exclusive or complete; ii) it is unlikely that all countries would apply all candidate activities; and iii) the analysis does not presume to reflect the final interpretations of Article 3.4. of the Kyoto protocol.

5.4 Additional References/Studies to Review:

- Review long-term forest re-growth rates. See: Lewis et al. (2009) (updated in RFS2) and/or Keith et al. 2009[7]
- Review regional crop and pasture modeling (such as updated by RFS2) for Brazil.

[7] http://www.pnas.org/content/106/28/11635.full

- Review international crop residue burning emissions.
- Consider international fertilizer production, livestock changes, enteric fermentation and manure management. Review work of SISC group[8] on EF for manure management.

There are many other factors, beyond agronomic factors, that limit land mobility within an AEZ, e.g., costs of conversion, managerial inertia, and unmeasured benefits from crop rotation.[9] At present, emission factors cannot capture how these activities will impact emissions from land uses projected past the baseline.

5.5 Recommendation

Short term

- Review assumptions of management practice effects on overall EF; particularly fire and "full tillage".
- Incorporate new data on perennial storage in roots, soil and aboveground biomass.
- Compile new data on disturbance effects and stages of sequestration at regional level.

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[8] Stewardship Index for Specialty Crops (SISC). here: http://www.stewardshipindex.org/

[9] Golub 2010 GTAP Paper: "Modeling Biofuels Policies in General Equilibrium: Insights, Pitfalls and Opportunities (Golub, Hertel, Taheripour and Tyner)

6. Long-term Carbon Storage in Harvested Wood Products (HWPs)

When biomass is harvested, some is left onsite and only a portion of forest biomass is removed from the forest. The biomass left onsite, typically non-stemwood tissues including the bark, branches and leaves, provides critical ecological services such as conservation and protection of soil health. The biomass removed from the forest (so-called merchantable biomass or harvested wood products, HWPs) usually ends up in one of four carbon pools: products-in-use, landfill, emitted with energy capture, emitted without energy capture. Products-in-use typically include industrial, commercial, and residential wood products such as construction materials (lumber, plywood, oriented strandboard, nonstructural panels) and miscellaneous products and paper. These wood products have different lifespans that typically range from 2–100 years, with longer lifespan for lumber and shortest for paper. The carbon disposition between these four pools will change over time as product-in-use move to landfills, where the carbon may be stored in the landfill or, upon decomposition, be released to the atmosphere as CO_2 or CH_4. Figure 6.1 illustrates the typical disposition of HWP by timber type and by products.

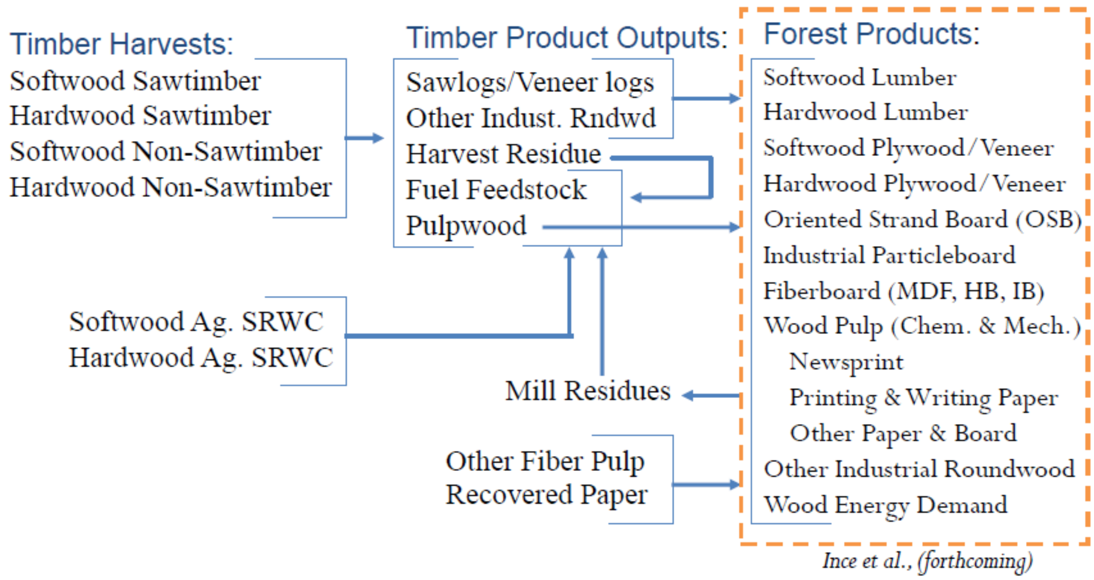

Figure 6.1: Disposition of harvested wood products. Source: Earles and Halog (2010).

The treatment of carbon stored in HWPs varies across models. For example, IPCC Tier 1 recommends treating HWPs as *instantaneous emissions* at the year of biomass removal from the forest system, it recommends that the storage of carbon in forest products be included in a national inventory only in the case where a country can document that existing stocks of long term forest products are in fact increasing (Grêt-Regamey et al. 2008). On the other hand, several other guidelines consider the storage factor and the fate of the carbon in HWPs, such as California Climate Action Registry (CAR 2009) and Chicago Climate Exchange dealing with project level carbon certification for managed forest, and government guidelines by U.S. DOE 1605 Forestry Emission Guidelines (U.S. DOE 2006) and USFS (Smith et al. 2006).

The CARB analysis assumes that when land is cleared, 100% of carbon in biomass (75% of forest biomass and 100% of grassland vegetation in the Tyner analysis) will be released into the atmosphere at the time of land conversion. We recommend that the inclusion of HWP pool in the ILUC analysis will better reflect the trimming of emissions. This will also allow for better accounting of substitution of energy intensive materials within a consequential LCA framework.

6.1 Carbon Stored in Wood Products

The calculation of carbon storage in HWPs requires the following key information:

- The (dynamic) market share of the HWPs over time
- The lifetime (fate of carbon stored) of HWPs over time

Figure 6.1 shows the disposition of forest biomass once it is disturbed ($t = 0$) for Canada (Wood and Layzell 2003), and the rest are belowground biomass and non-merchantable biomass that is left on the ground. Note that the biomass C values reported in WHRC include both aboveground and belowground biomass, typically using a 20% adjustment factor to account for belowground biomass C values if only aboveground biomass C values were available. As shown, only about 30% of the total forest biomass is in the merchantable wood, and only about 10% becomes lumber, the wood product category with the longest lifetime (Figure 6.1).

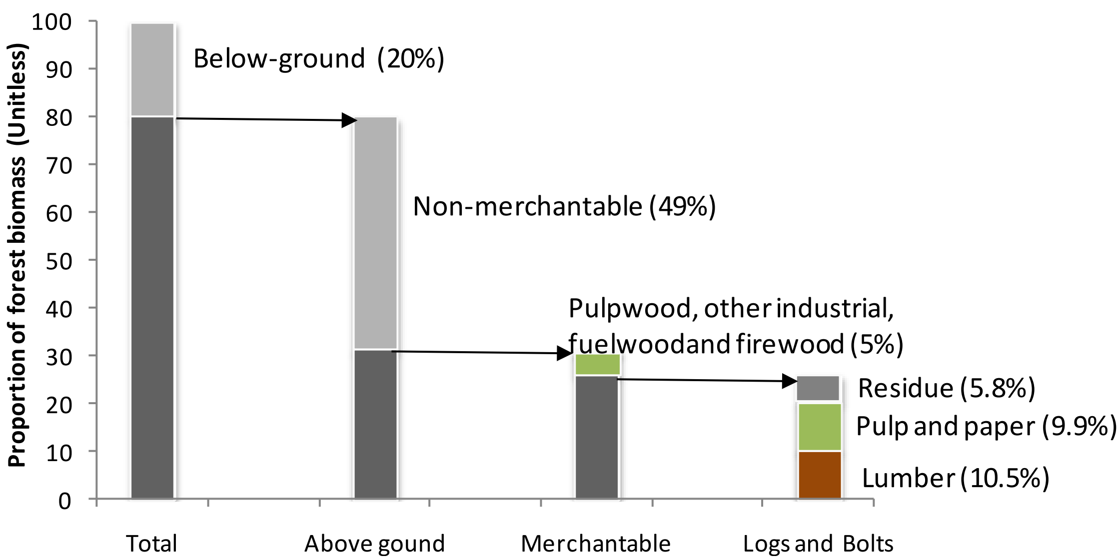

Figure 6.1: Disposition of forest biomass after disturbance. Source: (Wood and Layzell 2003).

The fate of wood products can be represented by decay functions of separate product pools, such as the ones shown below (Marland, Stellar, and Marland 2010).

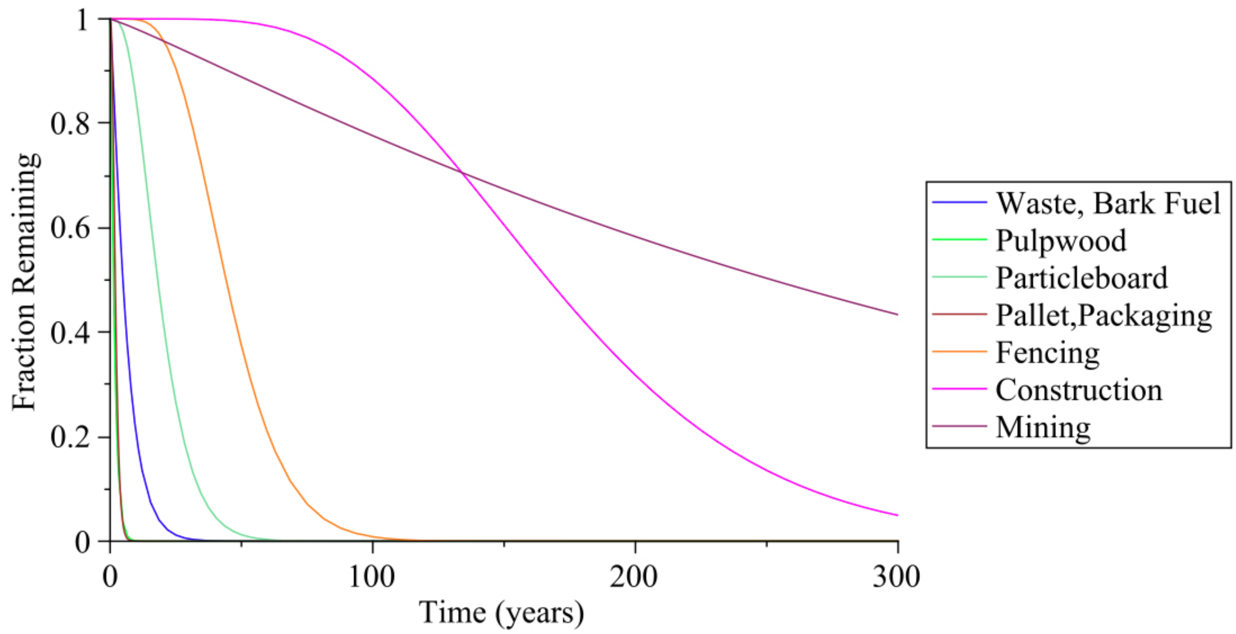

Figure 6.2: Fraction of carbon in primary wood products remaining in end uses after production. Source: (Marland, Stellar, and Marland 2010).

Using the information in Figure 6.1 and a similar information represented in Figure 2 but from the US source (U.S. DOE 2006), the total carbon stored in wood products is estimated to be 5.6%, 3% and 1.6% after 20, 50, and 100 years (Table 6.1).

Table 6.1: Fraction of carbon stored in end-uses after 20, 50, and 100 years by end-uses category. Source: Modified from Yeh et al. (2010).

Year after production	Softwood and hardwood lumber (%)	Pulp and paper (%)	Misc (%)	Total carbon stored in wood products (%)
20	5.3	0.13	0.20	5.6
50	3.0	-	0.04	3.0
100	1.6	-	0.00	1.6

The results shown here are significantly different from the discussion in Section 6.2. This calculation relies on the calculation of carbon remains in end-use products, while the calculation in Section 6.2 relies on tracking carbon stored by wood types.

The issue of HWP is also examined in the Winrock report for the RFS2 analysis (Harris et al. 2009, which concludes that "carbon stored in wood products long-term is probably immaterial for most regions of the world, especially if considering a timeframe of 30 years."

There are many accessible, international data on the production and trade of major categories of primary wood products that can be used for the HWP calculation, such as the ForesStat (http://www.fao.org/forestry/databases/29420/en/) or the FRA data (http://www.fao.org/forestry/32046/en/) by the Food and Agriculture Organization of the United Nations (FAO). Note that there are discrepancy of terminology, collection methods, and the quality of the data between databases. The quality of HWP data in the FAO database is reported to be variable e.g. +/- 10-15% for OECD countries and as high as +/-50% for non-OECD countries (Pingoud et al., 2003). Table 6.2 shows wood removals by country in 2005 and their disposition into fuelwood and industrial roundwood categories (FAO 2005). Data on major product categories (industrial roundwood, wood fuel, sawnwood, wood-based panel, wood pulp, paper and paperbord, recovered paper) by country is available for ForesStat. However, the disposition patterns of forest biomass removed due to land conversion may be different from those of commercially managed forests. In addition, if the mode of clearing is predominantly fire in some regions, then all of the biomass carbon will be combusted when LUC occurs.

Table 6.2: Wood removals 2005. Source: (FAO 2005).

Region/subregion	Industrial roundwood	Fuelwood		Total removals
	million m³	million m³	% of total	million m³
Eastern and Southern Africa	34	151	82	185
Northern Africa	8	173	96	181
Western and Central Africa	36	267	88	303
Total Africa	**79**	**591**	**88**	**670**
East Asia	115	56	33	171
South and Southeast Asia	44	113	72	157
Western and Central Asia	15	20	57	34
Total Asia	**174**	**189**	**52**	**362**
Total Europe	**543**	**139**	**20**	**681**
Caribbean	4	16	82	19
Central America	4	40	90	45
North America	717	56	7	773
Total North and Central America	**725**	**112**	**13**	**837**
Total Oceania	**54**	**10**	**15**	**64**
Total South America	**225**	**173**	**44**	**398**
World	**1 799**	**1 214**	**40**	**3 013**

6.2 Analysis of Carbon Release from Conversion of US Forest Ecosystems

Mueller (2010) combined data from USDA's Resource Planning Act RPA tables with a report on harvested carbon estimates from USDA (USDA 2006 Report).[10,11] The Mueller analysis did not

[10] Forest Inventory and Analysis National Program, 2007 RPA Resource Tables, available at http://www.fia.fs.fed.us/program-features/rpa

include wood combusted with energy capture or the displacement of wood products in use. Including these would have required a life cycle analysis with a displacement approach similar to the analysis performed by Oneil and Lippke (2009) or Scharai-Rad (2002).[12,13] A complete LCA would have required substantial time and resources. Mueller concludes that the attached results are likely underestimating the total harvested carbon. The derived factor supports the HWP factor used in Tyner et al. 2010 for the United States. It also confirms the importance of taking HWP into consideration. The analysis presented below employed the following steps:

1. The fractions of softwood and hardwood removals that leave harvest sites from forest ecosystems by region were determined using Table 40 from the RPA tables.[14] The derived values are listed in Table 6.3. Wood leaving harvest sites is termed harvested wood.

Table 6.3: Fractions of total wood removals that are softwood (or hardwood) that leave the harvest site and the fractions of softwood (hardwood) that leaves the site that is sawlogs or pulpwood.

	Fraction [cf/cf] of Useful Wood Removed (from RPA Table 40)		Fraction [cf/cf] End Use of Removals (From RPA Table 39)			
	softwood	hardwood	softwood sawlogs	softwood pulpwood	hardwood sawlogs	hardwood pulpwood
North East	0.21	0.47	0.56	0.39	0.39	0.31
North Central	0.10	0.50	0.37	0.60	0.36	0.46
Southeast	0.51	0.19	0.49	0.51	0.37	0.43
Pacific Northwest	0.73	0.05	0.87	0.07	0.63	0.32
Pacific Southwest	0.80	0.01	0.68	0.00	0.05	0.30
United States	0.47	0.24	0.64	0.31	0.40	0.41

2. The end use of the harvested wood was determined. Using RPA Table 39, the fractions of harvested softwood and hardwood that go into sawlogs and pulpwood were determined. Veneer logs and sawlogs were classified as sawlogs, whereas composite products, posts, poles, pilings, and miscellaneous products were classified as pulpwood.[15] The derived values are also listed in Table 6.3.

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[11] Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States. Prepared by James Smith, Linda Heat, Kenneth Skog, and Richard Birdsey, General Technical Report NE-343, April 2006. Available at http://www.treesearch.fs.fed.us/pubs/22954

[12] Oneil, Elaine and Bruce Lippke. "Life Cycle Carbon Tracking for the Working Forests of British Columbia: Carbon Pool Interactions from Forests, to Building Products, and Displacement of Fossil Emissions"; University of Washington, 2009.

[13] Scharai-Rad, Mohammad. "Environmental and energy balances of wood products and substitutes." University of Hamburg, Department of Wood Technology and Dr Johannes Welling Federal Research Centre for Forestry and Forest Products, Hamburg; 2002

[14]RPA Table 40 is titled "Roundwood products, logging residues, and other removals from growing stock and other sources by species group, region, and subregion, 2006." Forest Inventory and Analysis National Program, 2007 RPA Resource Tables, available at http://www.fia.fs.fed.us/program-features/rpa

[15] RPA Table 39 is titled "Volume of roundwood products harvested in the United States by source of material, species group, region, subregion, and product, 2006."

3. Now that the fractions of sawlogs and pulpwood that are produced have been identified as well as the fractions of wood that leave forest ecosystems, the disposition patterns of carbon for the following categories were determined: products in use and material in landfills (combined). The fractions for each year after production are provided in Table 6 of the USDA 2006 report. For the present study, the disposition fractions for the 30, 50, and 100-year time horizons were selected. The results are listed in Table 6.4.

Table 6.4: Disposition Patterns

	Disposition Patterns: 30 Years (from USDA 2006 Report Table 6)			
	softwood sawlogs	softwood pulpwood	hardwood sawlogs	hardwood pulpwood
North East	0.40	0.12	0.40	0.33
North Central	0.44	0.13	0.37	0.38
Southeast	0.43	0.18	0.38	0.24
Pacific Northwest	0.52	0.10	0.27	0.27
Pacific Southwest	0.45	0.45	0.45	0.45
United States	0.45	0.20	0.37	0.33

	Disposition Patterns: 50 Years			
	softwood sawlogs	softwood pulpwood	hardwood sawlogs	hardwood pulpwood
North East	0.36	0.10	0.36	0.30
North Central	0.39	0.11	0.33	0.35
Southeast	0.39	0.16	0.34	0.22
Pacific Northwest	0.47	0.09	0.24	0.24
Pacific Southwest	0.41	0.41	0.41	0.41
United States	0.40	0.17	0.33	0.30

	Disposition Patterns: 100 Years			
	softwood sawlogs	softwood pulpwood	hardwood sawlogs	hardwood pulpwood
North East	0.32	0.09	0.32	0.26
North Central	0.35	0.09	0.30	0.30
Southeast	0.34	0.14	0.30	0.19
Pacific Northwest	0.41	0.08	0.21	0.21
Pacific Southwest	0.36	0.36	0.36	0.36
United States	0.35	0.15	0.30	0.26

4. The fractions of wood removals (wood cut down) that are harvested wood were multiplied by the fractions used for sawlogs and pulpwood and the respective disposition fractions.

The results are listed in Table 6.5 below. For example, after 30 years, on average, 2 percent of carbon from the softwood of stands removed from forests in the North Central region has been allocated to products-in-use or landfilled. In addition, 16 percent of the carbon from the hardwood removed from forests in the North Central region has been allocated to products-in-use and landfills. In total, 18 percent of the carbon from North Central stands can be considered sequestered 30 years after harvest. Figure 6.3 shows the carbon sequestered by region for the three selected time horizons. Across the US, 23 percent of the carbon in harvested forest stands has been sequestered to products-in-use or landfills after 30 years, implying that 77% of all carbon has been released. Only slightly less carbon has been sequestered after 50 years and 100 years.

Table 6.5: Fractions of Carbon Sequestered

	30 years			50 years			100 years		
	softwood	hardwood	Total	softwood	hardwood	Total	softwood	hardwood	Total
North East	0.06	0.12	0.18	0.05	0.11	0.16	0.04	0.10	0.14
North Central	0.02	0.16	0.18	0.02	0.14	0.16	0.02	0.12	0.14
Southeast	0.15	0.05	0.20	0.14	0.04	0.18	0.12	0.04	0.16
Pacific Northwest	0.34	0.01	0.35	0.30	0.01	0.31	0.26	0.01	0.27
Pacific Southwest	0.24	0.00	0.25	0.22	0.00	0.22	0.19	0.00	0.19
Jnited States	0.16	0.07	0.23	0.15	0.06	0.21	0.13	0.05	0.18

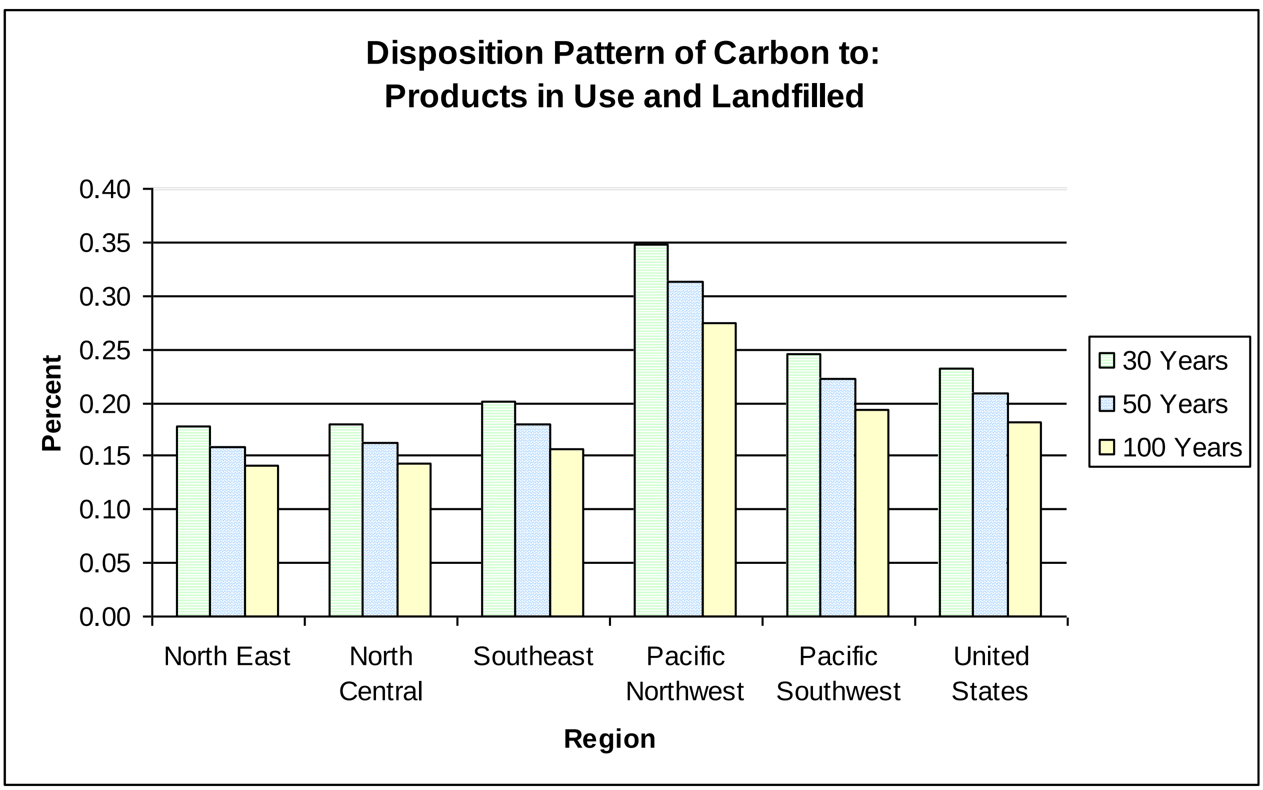

Figure 6.3. Carbon Sequestered by Region and Time. Source: Mueller (2010).

6.3 Other Factors Not Considered Above

The above discussion and examples do not take into account wood combusted with energy capture as well as the displacement of fossil-fuel intensive construction materials, both of which can displace GHG emissions from the use of fossil fuel. This would have required a life cycle analysis with a displacement approach similar to the analysis performed by Oneil and Lippke (2009) or Scharai-Rad (2002).[16,17]

6.4 Recommendations

Based on our examination, there is sufficient data to consider C storage in HWP in the US and other developed countries. Data for global HWP disposition by country and long-term carbon storage factor by wood-type and end-product is available but highly uncertain.

Short term

We recommend that the inclusion of the HWP pool in the ILUC analysis to better reflect the timing of emissions. Confidence intervals should be included for all parameters in the HWP storage model to allow them to be included in an uncertainty analysis.

Longer term

Consider consequential LCA by including the substitution by HWP for fossil energy and energy intensive materials as well as including energy used to produce wood products.

[16] Oneil, Elaine and Bruce Lippke. "Life Cycle Carbon Tracking for the Working Forests of British Columbia: Carbon Pool Interactions from Forests, to Building Products, and Displacement of Fossil Emissions"; University of Washington, 2009.

[17] "Environmental and energy balances of wood products and substitutes." Dr Mohammad Scharai-Rad University of Hamburg, Department of Wood Technology and Dr Johannes Welling Federal Research Centre for Forestry and Forest Products, Hamburg; 2002.

7. Other Non-Land Conversion Emissions

The Energy Independence and Security Act of 2007 (EISA) defined life cycle GHG emissions to include "direct emissions and significant indirect emissions such as significant emissions from land use changes" (United States Congress 2007). This definition, which was subsequently adopted verbatim into the California LCFS (OAL 2010), indicates that other indirect GHG emissions besides those from land-use change should be counted in each fuel's life cycle.

In its analysis for the Renewable Fuel Standard (RFS2), US EPA estimated changes in indirect GHG emissions (and sequestration) from the following categories:

1. Land clearing and conversion (above- and belowground biomass, soil carbon, forgone sequestration)
2. Tillage (conversion to no-till agriculture is awarded carbon sequestration credit)
3. Fertilizer and on-farm energy use (i.e., emissions are added or subtracted based on average practices in each country where changes in crop production occur)
4. Methane from rice and livestock production

In its current analysis, CARB considers only item 1, ILUC emissions. Items 2 through 4 are discussed in the next sections.

7.1 Emissions Associated with Market Mediated Effects

To estimate market-mediated effects, the GTAP model is "shocked" using a combination of a subsidy for biofuels and a revenue-neutral tax on transport fuels more generally. The model seeks a new equilibrium in which corn ethanol production increases and other commodities and sectors adjust to new prices by increasing or reducing production and consumption.

These changes in production can result in significant changes in GHG emissions, for example from:

- Livestock emissions
- Rice production emissions
- Crop switching

The changes can be positive or negative and can differ regionally. Looking at the EPA RFS2 analysis, which did attempt to calculate these emissions, the results were significant for some biofuels (~25% of ILUC for soybean biodiesel) and very small for other fuels.

7.1.1 Livestock Emission Factors

Agricultural emissions accounted for about 32% of total US CO_2-equivalent (GWP_{100} basis) anthropogenic emissions in 2000. Livestock emissions (including indirect emissions) account for about 42% of these emissions in two major categories:

- Enteric fermentation (~34% of total ag emissions)
- Manure (~8% of total ag emissions). Indirect emissions from manure management are also highly variable and substantial.

Changes in livestock population directly impact both of these emission sources. The results from the EPA analysis for changes in livestock emissions are shown in the following table.

	g CO_2eq/MJ
Corn Ethanol	-0.27
Soybean Biodiesel	-8.07
Sugar Cane Ethanol	-0.12

It should be possible to extract changes in livestock populations from the GTAP model runs. Using this data and regional emissions derived from the EPA[18], or from the UNFCCC Annex 1 countries; it is possible to calculate the indirect GHG emissions from changes in livestock populations.

7.1.2 Rice Cultivation

There is a similar situation for rice production emissions. Rice emissions account for 11% of agricultural emissions. EPA's estimates of the emission impacts of changes in rice cultivation by biofuel are shown in the following table.

	g CO_2eq/MJ
Corn Ethanol	1.78
Soybean Biodiesel	-5.45
Sugar Cane Ethanol	0.46

Same basic approach would be taken to quantifying these emission changes. Take the change in rice production from GTAP, multiply by the rice emissions per GTAP region. These can be obtained from the EPA or UN FCCC inventories for Annex 1 countries.

7.1.3 Crop Switch

A more complex issue is the change in emissions from land that remains in the same land use category, such as cropland remaining cropland. GHG emissions per acre are significantly different between crops, rotation and management practices as shown in the following Figure 7.1[19].

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[18] US EPA. Global Anthropogenic Non-CO_2 Greenhouse Gas Emissions: 1990-2020. http://www.epa.gov/climatechange/economics/downloads/GlobalAnthroEmissionsReport.pdf

[19] Dyer JA, et al, The impact of increased biodiesel production on the greenhouse gas emissions from field crops in Canada, Energy for Sustainable Development (2010), doi:10.1016/j.esd.2010.03.001

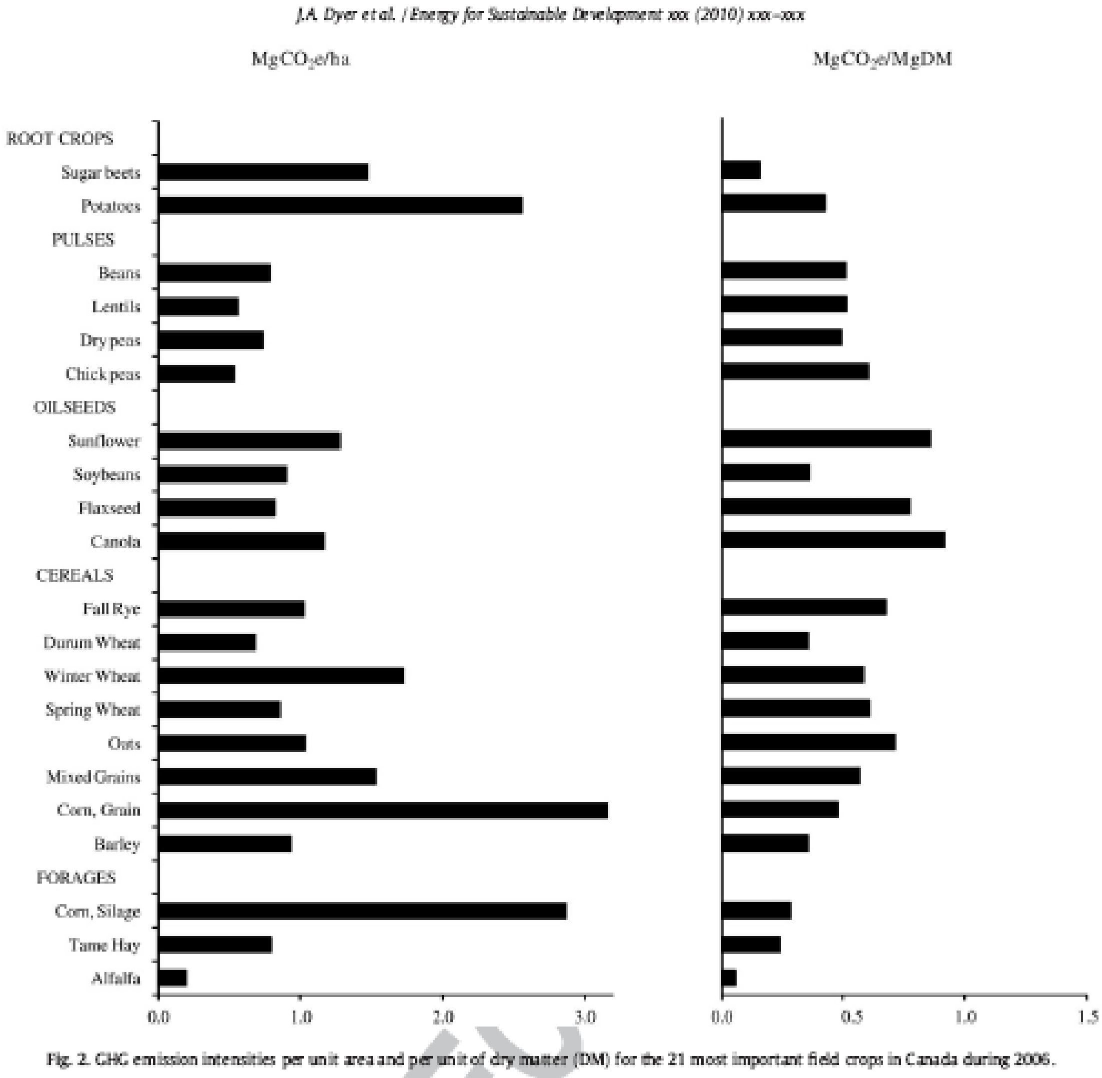

Figure 7.1: GHG emission increases per unit area and per unit of dry matter (DM) of the 21 most important field crops in Canada during 2006.

The total GHG emissions from cropland therefore depend on the crop mix and field management practices. The assumption that has effectively been made, that there are no GHG impacts of cropland remaining cropland, is obviously not correct. The issue is that some of the crop shifting is driven by the availability of co-products, whereas other crop shifting is caused by demand changes resulting from changes in prices. In the direct GHG analysis, we already attempted to put a GHG values on those co-products, so there is some overlap between the GHG change from crop shifting and the GHG benefits from the direct analysis of co-products.

It is not clear how this could be resolved in the short term. An additional complexity is that GTAP probably does not provide enough detail on the various crops, given its sectoral aggregation, e.g., barley and corn are aggregated into coarse grains, but have very different emissions per acre.

7.2 Emissions Changes Associated with Agricultural Practices

In the RFS2 analysis, EPA projected changes in emissions from fertilizer use, on-farm energy use, and production of rice and livestock, both domestically and internationally. Domestically, EPA relied on FASOM for these projections, while internationally, cropping changes were combined with estimates of average input and emission rates for each country.

These same two approaches are available to CARB for use with GTAP. GHG emission factors are available for use in GTAP, both for CO_2 and non-CO_2 emissions[20] allowing the model to estimate marginal GHG changes that are consistent with estimates of ILUC emissions. (we note that while FASOM has much finer resolution, it also has much more limited scope, being a domestic, partial equilibrium model; neither model is ideal for the task at hand.)

7.2.1 Tillage Changes

In its modeling for RFS2, the US EPA combined two economic models: FASOM was used to model the US, and FAPRI was used for the rest of the world. The FASOM model estimates GHGs associated with agriculture and forestry, including soil carbon sequestration assumed to be associated with a transition from conventional tillage (CT) to no-tillage (NT). FASOM also projects changes in carbon stocks associated with forestry, though these projections include large unexplained anomalies (Plevin et al. 2010a).

Experimental data performed by several researchers indicate that in some cases carbon sequestration can potentially occur due to changes in field management practices (e.g., tillage) especially when switching from intensive tillage methods to no-till. Carbon sequestration rate is dependent upon a number of factors including crop, yield, rotation, tillage practices and timing, climatic effects, etc. One set of long-term field experimental data (West and Post 2002) indicate carbon sequestration rates have increased when changing from conventional tillage to no-till field management practices especially in the 0-30 cm soil depth (~1 foot). Work by Angers and Erikson-Hamel (2008) showed significant differences in soil organic carbon (SOC) stocks between intensive tillage systems and no-till situations occur at the soil surface but also at depth, which further highlights the importance of taking into account the whole soil profile when comparing soil C stocks.

However, others have found that the assumed carbon sequestration benefits of no-tillage over conventional tillage may only occur at the shallow depths (less than 30 cm) (Baker et al. 2007; Batlle-Bayer, Batjes, and Bindraban 2010; Gál et al. 2007; Luo, Wang, and Sun 2010; Yang et al. 2008). Baker et al. (2007) even concluded that while "there are other good reasons to use

[20] See https://www.gtap.agecon.purdue.edu/models/energy/Land_Use/

conservation tillage, evidence that it promotes C sequestration is not compelling." Van den Bygaart and Angers (2006) found high variability in SOC stocks often makes it difficult to detect differences induced by management practices and at greater depths the effect of tillage (field management). Intensive tillage (sometimes full inversion) and no-till have been shown in limited experimentation to result in different redistributions of SOC in the soil profile and consequently, the net effect of switching tillage practices to no-till on total C stocks is difficult to predict. Much is still unknown at this point concerning actual carbon sequestration in soils and its effect on climate change.

FASOM incorporates from DAYCENT emission factors for soil GHG fluxes. The specifics of these factors and where exactly they are applied have not been documented by USEPA. Page: 41 The impact of reduced tillage on N_2O emissions is a function of soil type and climatic conditions. Basically, conditions that promote high N_2O emissions (high moisture and saturated soils) will produce a short-term increase in N_2O emissions. Soils that have low N_2O emissions can see a decrease in N_2O with a switch to no tillage. Six et al. (2004) concluded that conversion to no-till can increase N_2O emissions for decades, resulting in a net increase in global warming potential—even assuming no-tillage results in carbon sequestration—yet it's unclear whether FASOM accounts for this. The large CO_2e benefit FASOM assigns to conversion to no-till seems to indicate that this N_2O increase is not included. As far as we know, EPA did not account for tillage changes outside the US.

GTAP doesn't represent tillage and therefore is silent on the matter. However, given the recent findings challenging the soil carbon benefits of reduced tillage, we do not recommend including this effect at this time. However, in the context of an uncertainty analysis, it would be appropriate to represent a range of outcomes based on variation in assumptions about the incidence of tillage reduction and the resulting changes in soil carbon.

Finally, it is important to look at consistent depth levels between soil carbon releases during land conversion and subsequent carbon sequestration from till/no till practices. In general, the assessed soil carbon depth should be the same as the assessed till/no till depth.

7.2.2 Fertilizer Use and N_2O Emissions

N_2O emissions from soils are an important part of the lifecycle GHG emissions for all biofuels. Most analyses of the issue utilize at least part of the methodology recommended by the IPCC. Recently, Crutzen et al.[21] suggested that the IPCC methodology may significantly underestimate N_2O emissions by a factor of three to five and that as a result, the lifecycle GHG emissions of biofuels may be greater than those of petroleum fuels.

It is important to understand that the methodology employed by Crutzen and that of the IPCC are very different and cannot be directly compared. The Crutzen approach has been described as a "top down" method and the IPCC is very much a bottom-up approach.

[21] PJ Crutzen *et al, Atmos. Chem. Phys. Discuss*., 2007, **7**, 11191

The Crutzen approach estimated global N_2O emissions from the atmospheric concentrations of N_2O and estimated the portion that was attributable to agricultural soils by eliminating the estimated contributions from other sources. This was then compared to nitrogen fertilizer application rates to arrive at a value of about 3.4 to 4.6%, with the range resulting from uncertainty in the individual values. This approach is very sensitive to the accuracy of the values being eliminated. It has been suggested that the Crutzen paper missed some sources such as biomass combustion, livestock, and even transportation (Mortimer et al)[22]. The range would be only 2.8 - 4.2% if these additional sources were included.

The IPCC bottom-up approach is based on field measurements of N_2O from a large number of studies around the world. The IPCC Tier 1 value of 1% is also widely misunderstood, as this is the average for the direct emissions only. The IPCC also has estimates for indirect effects and for emissions at later stages of the lifecycle. For example, when straw that is also produced with grain or oilseeds is returned to the soil, it contains nitrogen that results in N_2O emissions that are not included in the original estimate of 1%. The next year's crop would use a portion of this nitrogen, and the straw from that crop would also decompose and release some N_2O, and so on. If the feedstock is used to feed an animal there are additional emissions of the nitrogen when the manure from the animal is returned to the soil, and so on through the cycle. There are also indirect emissions when some of the nitrogen is leached from the soil.

When the total soil N_2O emissions from the IPCC approach are calculated, the range of values produced 0.6 - 3.5%, which has some overlap with the Crutzen values. It is widely recognized by soil scientists that the N_2O emission factor is a function of soil composition and climatic conditions. Many countries, including Canada and the United States, use a more detailed approach to estimating N_2O emissions from agricultural soils than the IPCC Tier I approach. In Canada, this more detailed approach results in direct emission factors in the range of 0.5 to 1.7% and a total emission factor of 3.8%. Lifecycle studies that are well done will use N_2O emission factors that are appropriate for the region and system being modelled.

Many biofuel pathways co-produce animal feed, requiring that some of the emissions be attributed to the co-product. If the GHG emissions increase for producing a crop displaced by a co-product, then the GHG emissions avoided by the use of this substitute animal feed also increase. Thus there is not necessarily a direct relationship between the total lifecycle emissions and the N_2O emission rate from fertilizer application.

7.3 Recommendations

Short term

[22] N.D. Mortimer, A. Ashley, A. Evans, A. J. Hunter and V. L. Shaw. 2008. Support for the Review of the Indirect Effects of Biofuels. http://www.globalbioenergy.org/uploads/media/0806_North_Energy_-_Support_for_the_review_of_the_indirect_effects_of_biofuels.pdf

- Consider a broader range of significant indirect emissions from land use changes such as, but not limited to, those related to livestock and rice production.

Longer Term

- Consider the complexity of GHG emissions from crop switching

8 Non-Kyoto Climate Active GHG and Aerosol Emissions

LCA typically considers only one category of climate effects—direct greenhouse gases. Within this category, LCA studies generally consider at most six gases (or groups of gases) defined in the Kyoto Protocol as contributing to global warming effects: CO_2, methane (CH_4), nitrous oxide (N_2O), hydroflourocarbons (HFC), perflourocarbons (PFC), and sulfur hexafluoride (SF_6). Fuel cycle models typically consider only the first three gases, on the presumption that little or no HFC, PFC, or SF_6 is emitted in the life cycle of transportation fuels (USEPA, 2009b, p. 302). The standard approach for aggregating climate effects is to sum the emissions of these "big three" gases—CO_2, CH_4, and N_2O—weighted by the latest IPCC global warming potential values (e.g., Forster et al. 2007) using a 100-year time horizon (CARB 2009; USEPA 2010; Wang 1999, 2009).

However, several other compounds emitted over fuel life cycles are climate-active: carbon monoxide (CO), non-methanol volatile organic compounds (NMVOC), sulfur oxides (SO_X) oxides of nitrogen (NO_X), black carbon (BC), and organic carbon (OC) all affect climate, though their global warming effects are in some cases variable over time, space, and chemical conditions, and in general, uncertain (Delucchi 2003; Forster et al. 2007; Kammen et al. 2007; Larson 2006; Sanhueza 2009). Despite the variability and uncertainty, an important question is whether inclusion of these emissions has the potential to alter the preference order of alternative fuels with respect to their effects on climate. Table 8.1 lists CO_2-equivalent global warming potentials for the three well-mixed GHGs and shorter-lived species.

Table 8.1: Sample GWP_{20} and GWP_{100} CO_2 equivalence factors for various substances.

Substance	GWP_{20}	GWP_{100}
CO_2	1 [a]	1 [a]
CH_4	72 [a]	25 [a]
N_2O	289 [a]	298 [a]
CO	10 [b]	3 [b]
SO_2	-140 [e]	-40 [e] -94 [c]
NO_X		-1 [c]
NMVOC	14 [e]	4.5 [e] 8 [c]
BC	1600[e] 2200 [d]	460[e] 680 [d]
OC	-240 [e]	-69 [e]

[a] Forster, Ramaswamy et al. (2007)
[b] Sanhueza (2009) (CO GWPs are for sustained releases, based on Fuglestvedt et al. (1996))
[c] Brakkee, Huijbregts et al.(2008)
[d] Bond and Sun (2005)
[e] Fuglestvedt, et al. (2010). Global average are presented though regional values will vary. The effect on clouds (and in the case of BC, surface albedo) is not included.

8.1 The Role of Non-CO_2 Emissions in Land Clearing Emission Factors

The mode of clearing (burning vs. smoldering vs. mechanical) affects the BC and CO emission factors. Table 8.2 shows emissions of trace gases and aerosols for savanna fires, in mass and as CO_2-equivalents. If only CO_2, CH_4, and N_2O are considered, the emission factor per kilogram of dry matter would be 1745 g CO_2e kg^{-1} (GWP_{100}) or 1856 g CO_2e (GWP_{20}). Inclusion of the remaining emissions shown in Table 8.2 and the GWP values shown in Table 8.1 increases these emission factors to 2346 g CO_2e kg^{-1} and 3648 g CO_2e kg^{-1}, respectively. Under a short time horizon (e.g. 20 years) black carbon can contribute more radiative forcing than the CO_2 released when burning biomass. In either timeframe, including all the emissions greatly increases the CO_2-equivalent emission factor. However, the much greater uncertainty in the global warming potential values must be taken into account.

Burning results in higher BC emission rates (shown in table), whereas smoldering (not shown) results in lower BC emissions but higher CO emissions. (Ward 1991 discusses the different types of fires and combustion efficiencies.)

Table 8.2: Emissions of trace gases and aerosols for savanna fires, in mass (g kg^{-1}) and as CO_2-equivalents (g CO_2e kg^{-1}).

		100-year GWP			20-year GWP		
Emission	**g/kg dm[a]**	**EF**	**g CO_2e/kg**	**Contribution**	**EF**	**g CO_2e/kg**	**Contribution**
CO_2	1640	1[b]	1640	70%	1[b]	1640	45%
CO	65	3[e]	195	8%	10[e]	650	18%
CH_4	2.4	25[b]	60	3%	72[b]	173	5%
NMHC	3.1	8[c]	25	1%	*8[c]*	25	1%
NO_X	3.1	-1[c]	-3	0%	*-1[c]*	-3	0%
N_2O	0.15	298[b]	45	2%	289[b]	43	1%
BC	0.8	680[d]	544	23%	2200[d]	1760	48%
OC	3.2	-50[e]	-160	-7%	-200[e]	-640[e]	-18%
Total			2346	100%		3648	100%

[a] Delmas, Lacaux et al. (1995)
[b] Forster, Ramaswamy et al. (2007)
[c] Brakkee, Huijbregts et al. (2008) – only 100-year GWP was reported.
[d] Bond and Sun (2005)
[e] Sanhueza (2009)

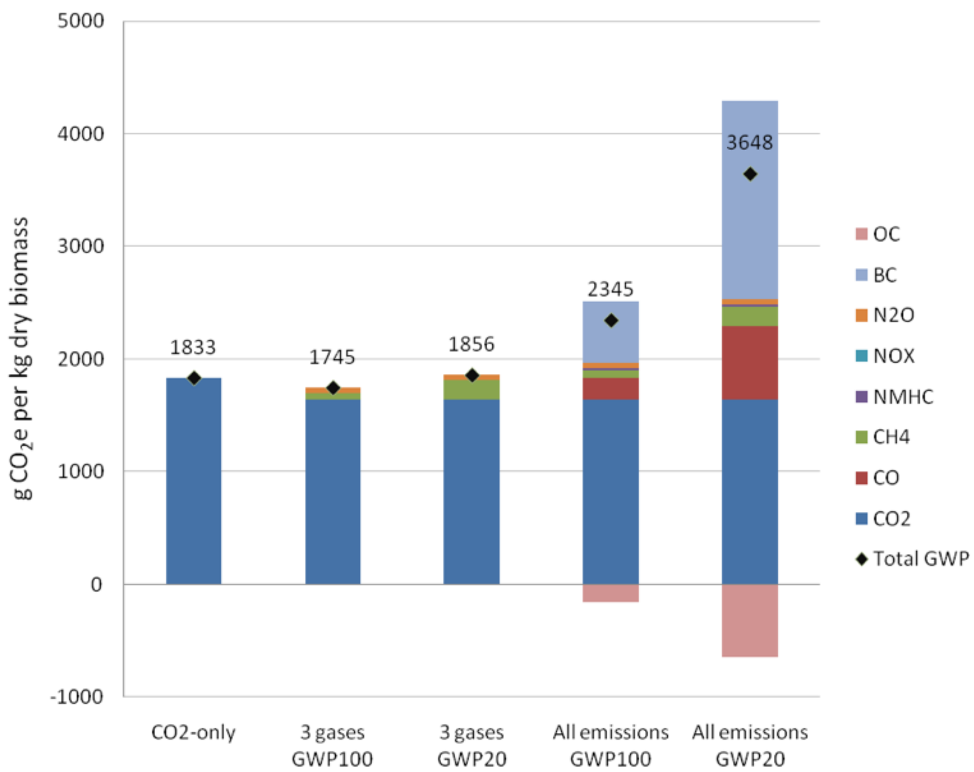

Figure 8.1: Savanna burning emission factor

Black carbon is emitted from land clearing, but also from the combustion of fuels including gasoline and diesel. In fact, Jacobson points out that considering black carbon emissions (under California's LEVII standards for PM at 0.01 g/mi) diesel would warm climate more than gasoline emissions (2002) on a full life cycle basis. Including black carbon as part of the LCFS analysis will a) increase the GWI for the gasoline and diesel reference fuels and b) alter the perceived environmental benefits between fuels.

8.2 Recommendations

Non-Kyoto climate forcing gases and particles can contribute to large uncertainties and significantly increase the estimated impacts of biofuel LUC emissions. The inclusion of these non-Kyoto emissions for ILUC would require their inclusion throughout the lifecycles of all fuels participating in the LCFS.

- Sensitivity analysis (short term) and uncertainty analysis (longer term) should be performed to explicitly consider the effects of non-Kyoto climate forcing gases and particles.

9 Uncertainty Analysis

Estimates of ILUC emissions require linking together various sets of uncertain data using imprecise models. To understand the uncertainty in the final estimate of ILUC emissions requires propagation of uncertainty through the combined economic-ecosystem model.

9.1 EPA's Uncertainty Analysis

In their regulatory impact analysis for RFS2, USEPA (2010) compared frequency distributions for biofuels with the required reduction thresholds, but these distributions included only uncertainties in the remote sensing and carbon accounting portions of the model. Figure 9.1 is taken from the Regulatory Impact Analysis for RFS2, showing a frequency distribution for the percentage reduction in corn ethanol GWI in 2022 versus the required 20% reduction threshold. This distribution is based on a Monte Carlo simulation that considers as uncertain remote sensing data (and change detection) and numerous parameters required to model emission factors. The analysis treats economic model output as certain. As a result, distributions such as that shown in this figure are not appropriate for the purpose of determining the probability of compliance.

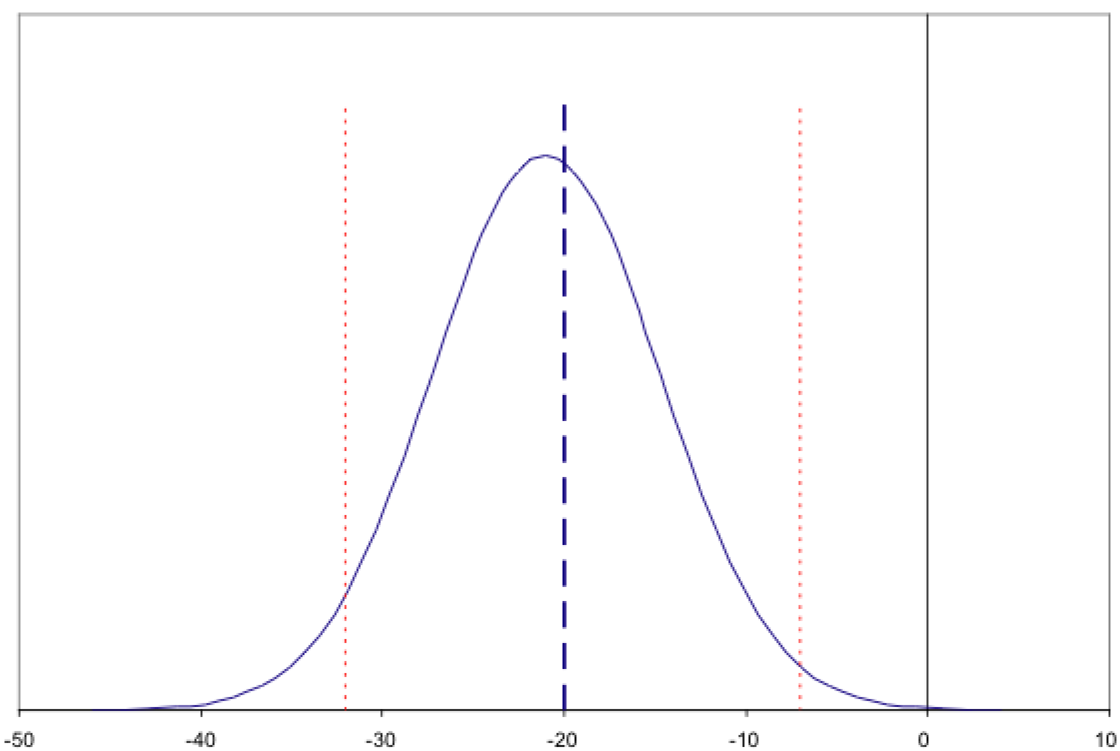

Figure 9.1: Distribution of 2022 corn ethanol GWI reduction relative to 2005 gasoline (for natural gas fired facilities producing 63% dry and 37% wet DGS, with fractionation). (Source: USEPA 2010, PDF p. 480).

Hertel et al. (2010a) incorporated into GTAP- probability distributions for the emission factors for each type of land conversion in each GTAP region and combined these with distributions around key elasticity parameters using GTAP's Systematic Sensitivity Analysis (SSA) feature. The details of this analysis are available in the supporting materials for that paper (Hertel et al. 2010b). The SSA is based on the Gaussian Quadrature method, which requires far fewer model runs than do Monte Carlo type methods, but SSA is subject to several important limitations. First, SSA assumes that input distributions are approximately normal. It's unclear whether this is true for economic model parameters, but it is almost certainly false for emission factors. Second, SSA produces only a mean and standard deviation as output, and thus is unable to represent skewed output distributions. Based on a reduced-form model of ILUC, Plevin et al. (2010b) generate output distributions that are all heavily skewed, with long right tails. Thus the SSA may underestimate both the median and breadth of the output distributions.

As a long-term goal, we recommend that CARB develop a stochastic analysis using a similarly unified model in which emission factor uncertainties are incorporated into GTAP. To enable this, we recommend developing a stochastic model combining remote sensing uncertainties with emission factor uncertainties, much as USEPA has done, to use in place of the distributions generated for the Hertel et al. analysis. In the near term, the distributions around emission factors can be used with GTAP's SSA, which, despite its limitation, is better than no analysis of uncertainty. In the long run, however, a more robust analysis would use Monte Carlo analysis to properly represent asymmetrical distributions.

9.2 Recommendations

Short term:

- Report all stock and EF model parameters with uncertainty ranges or intervals. Use these distributions (or symmetric distributions derived from them) in GTAP's Systematic Sensitivity Analysis (SSA) feature to estimate the mean and standard deviation of the ILUC emission factor considering the combined uncertainty in economic and carbon accounting parameters.

Longer term:

- Propagate uncertainty using Monte Carlo simulation.

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- Soil Carbon Stock: Sahoko Yui (lead author), Holly Gibbs, Sonia Yeh
- Peatlands Emissions: Sonia Yeh (lead author) and Sahoko Yui
- Emission Factors: Susan Tarka Sanchez
- Long-term Carbon Storage in Harvested Wood Products (HWPs): Sonia Yeh (general discussion) and Steffen Mueller (US forest products)
- Fertilizer Use and N_2O Emissions: Don O'Connor
- Other Non-Land Conversion Emissions: Don O'Connor (lead author), Richard Nelson, and Richard Plevin
- Non-Kyoto climate active GHG and aerosol emissions: Richard Plevin
- Uncertainty Analysis: Richard Plevin

Outside experts consulted:

- Gregg Marland, Oak Ridge National Laboratory and Mason Earles, University of Maine (Chapter 6)

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