APPENDIX C

Changes in carbon stocks related to land use and land-use change

1. INTRODUCTION

This appendix discusses GHG emissions and changes in carbon stocks that result from land use and LUC. Land uses and LUCs are defined; the relevant carbon pools and emission sources are discussed in the context of these categories; the approaches to estimating emissions and changes in carbon stocks are outlined; and finally, justification for and an explanation of the selected estimation methods used in this study is also provided.

Land use, LUC and forestry (LULUCF) is defined by the United Climate Change Secretariat as: *a greenhouse gas inventory sector that covers emissions and removals of greenhouse gases resulting from direct human-induced land use, land-use change and forestry activities*. Six land use categories are defined in the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: 1. Forest Land; 2. Cropland; 3. Grassland; 4. Wetlands; 5. Settlements; and 6. Other Land.

Land may remain in any of these categories or, in the case of LUC, its use may change to another category (e.g. from forest to grassland). Thus, each land use category can be further subdivided into land that is converted from one land use category to another, and land that remains in the same category. While this study focuses on the emissions from LUC, emissions from land use are also discussed.

1.1 GHG emissions from land-use change

Most LUCs alter the soil and vegetation of the land, thus changing the amount of carbon stored per unit area. These changes may be positive or negative, and may occur in each carbon pool: biomass (above- and below-ground); dead organic matter (dead wood and litter); and soil (soil organic matter).

LUC can significantly alter the carbon stored in biomass, by replacing the vegetation of the existing land use category with the vegetation of another land use category. Conversion of forest land to either grassland or cropland can lead to large and rapid losses of the typically large stores of carbon in forest vegetation, when this vegetation is replaced with herbaceous grasses or annual crops.

While most of the carbon stored in forest biomass is lost following conversion, some carbon will be transferred from one pool to another; e.g. when trees are felled, a portion of the above-ground biomass is transferred to the dead organic matter pool, and a portion of the below-ground biomass is transferred to the soil organic matter pool.

The drainage and cultivation or grazing of organic soils is also an important cause of the oxidation and loss of SOC for both croplands and grasslands (Armentano and Menges, 1986). While the most important GHG emission flux is CO_2, the oxidization of the various organic carbon pools as a consequence of LUC can also release N_2O.

Land conversion often results in an abrupt change where most biomass is lost, followed by a longer period where biomass is oxidized at a much slower pace. The IPCC (2006) assumes a default 20-year transition period following conversion over which all losses are accounted for.

The conversion of forest land to agricultural land may also lead to losses from the SOC pool. When forest land is converted to cropland, there is an average reduction in soil carbon of between 25 and 30 percent in the upper metre of soil (Houghton and Goodale, 2004).[14] These soil carbon losses are due, in part, to a lower fraction of non-soluble material in the more easily decomposed crop residues, and to the breaking up of aggregates and subsequent exposure of organo-mineral surfaces to decomposers following tillage (Post and Kwon, 2000). On the other hand, because grasslands, unlike crops, are not ploughed (temporary cultivated pastures are classified to be crops), little change in soil carbon is expected following the conversion of forests to grasslands (Houghton and Goodale, 2004).

When either cropland or grasslands are abandoned, there is a re-accumulation of carbon in vegetation as the land returns to its natural state, and the greater the biomass of the returning vegetation the larger is the long-term carbon sink due to the recovery. Post and Kwon (2000) note relatively low rates of accumulation in mineral soil following the abandonment of cropland. Considering all LUCs during the 1990s, Houghton & Goodale (2004) estimate that the average annual emissions from LUC were estimated to be 2.2 petagram C yr^{-1}, with almost all of this emanating from deforestation in the tropics.

1.2 Land use and its effects on emissions and carbon stocks

Agricultural lands hold substantial carbon stocks, mostly in soil organic matter. Carbon stock changes in agricultural lands are closely tied to management practices, which can either enhance or erode carbon stocks. Practices which raise (lower) the photosynthetic input of carbon and/or slow (accelerate) the release of stored carbon through respiration, erosion or fire will increase (decrease) carbon stocks (Smith *et al*., 2007). While it is possible for substantial biomass carbon to be stored through perennial plantings on agricultural lands (e.g. silvopastoral systems), carbon accumulation and losses occur mostly in the SOC pool. This below-ground carbon pool also has slower rates of turnover than above-ground pools, because most of the organic carbon in soils comes from the conversion of plant litter into more persistent organic compounds (Jones and Donnelly, 2004).

Smith *et al.* (2007) estimated that 89 percent of the agriculture sector's total mitigation potential is from SOC sequestration. For grasslands, practices such as the optimization of grazing intensities to maximize grass production, moderate intensification of nutrient-poor grasslands, and the restoration of degraded pastures are known to improve sequestration rates (Conant and Paustian, 2002; Sousanna *et al.*, 2010). Conversely, the overgrazing of grasslands reduces vegetation and the amount of litter returned to soils, and it leads to erosion and degradation contributing to CO_2 losses from the SOC pool. For croplands, significant changes in SOC stocks are associated with management practices including tillage, residue management, nutrient management and the use of organic amendments (Smith *et al.*, 2007).

[14] While there is some variation around this range, it has been documented in numerous studies, and has been found to be broadly robust across all ecosystems (Houghton and Goodale, 2004).

Historically, while agricultural management practices can result in either reductions or accumulations in the SOC pool, agricultural lands are estimated to have released more than 50 petagram C (Paustian *et al.*, 1998; Lal, 1999, 2004), some of which can be restored via better management. Currently, however, the net flux of CO_2 between the atmosphere and agricultural lands is estimated to be approximately balanced (Smith *et al.*, 2007). For the estimation of net livestock sector GHG emissions, which is the main purpose of this report, measures of net CO_2 current fluxes by region are of greater interest than the sequestration/mitigation potential.

The lack of a globally consistent and regionally detailed set of net CO_2 flux estimates make it difficult to quantify these potential emission sources and sinks by region in this study, although there are some relevant studies that provide useful estimates of these net fluxes for specific regions and agricultural land use categories. For example, based on literature observations for temperate grasslands mainly from Western Europe, Soussana *et al.* (2010) estimate that grasslands SOC sequestration rates averaged 5 ± 30 gC/m^2 per year. Nevertheless, Soussana *et al.* (2010) concede that the uncertainties associated with SOC stock changes following changes in management are very high. Further, stocks of SOC are very vulnerable to disturbances, including tillage, fire, erosion, and droughts that can lead to rapid reversals of accumulated stocks. Moreover, the authors recommend that further research is needed to separate the influence of management factors from other climate-related factors such as average temperature increase and CO_2 fertilization, in order to be able to attribute sequestration to direct anthropogenic causes.

There is also considerable potential to sequester carbon in croplands through a range of options available that include reduced and zero tillage, set-aside, perennial crops, deep rooting crops, more efficient use of organic amendments, improved rotations, irrigation, etc. In Brazil, for example, long-term field experiments (Costa de Campos *et al.*, 2011; Dieckow *et al.*, 2010; Vieira *et al.*, 2009; Sisti *et al.*, 2004) have evaluated the impact of conservation tillage and crop rotations on SOC. The results from these studies confirm that non-tillage and crop rotations can enhance the conservation of SOM and increase C accumulation. For example, Dieckow *et al.* (2010) who assessed the 17-year contribution of no-tillage crop rotations to C accumulation in subtropical Ferralsol of Brazil concluded that crop-forage systems and crop-based systems with legume represent viable strategies to increase soil organic C stocks. They found that the alfalfa system with maize at each three years showed the highest C accumulation (0.44 tonnes C/ha/yr). The bi-annual rotation of ryegrass (hay)-maize-ryegrass-soybean sequestered 0.32 tonnes C/ha/yr. However, an assessment of realistically achievable potentials for carbon sequestration in croplands needs to take into account economic, political and cultural constraints and other environmental impacts (such as non-CO_2 GHG emissions) also need to be accounted for.

2. QUANTIFICATION OF CARBON EMISSIONS AND SEQUESTRATION

2.1 Changes of carbon stocks related to land-use change

The most fundamental step in assessing emissions from LUC is the tracking of changes in areas of land use and conversions from one land use category to the next. This requires a time series of data, or at least two points in time, to capture changes in the area of land for each category.

Comprehensive guidance on methodological approaches for estimating LUCs as well as emissions and removals from LULUCF is provided in the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). Three different approaches are suggested with differing degrees of accuracy to best ensure the consistent representation of LUCs for given data quality and availability. The most accurate of these, Approach 3, requires the use of spatially-explicit data for land use categories and conversions, and includes the use of gridded map products derived from remote sensing imagery. At the other extreme is Approach 1, which relies on non-spatially explicit data from census and survey data, often reported at country or province level, and which only permits net changes in land use categories over time, and cannot specify inter-category conversions. Finally, Approach 2 enables the tracking of conversions between land use categories without the spatially-explicit location data. Naturally, the choice among the simple and more sophisticated approaches involve big trade-offs between the data and analytical resource requirements, and the accuracy with which LUCs and their attendant emissions and carbon removals are estimated.

For grassland remaining grassland, cropland remaining cropland, and conversion from forestland to either of these land use categories, the 2006 IPCC Guidelines require that changes in carbon stocks from each carbon pool (i.e. above-ground biomass, below-ground biomass, dead wood, litter and soil organic matter), as well as emissions of non-CO_2 gases, are estimated. The guidelines do, however, provide flexibility in the use of methods that range from very simple approaches that rely on default emission factors to more sophisticated approaches that use detailed location-specific data and process models that fully characterize the fluxes between carbon pools.

2.1.1 Biomass and dead organic matter (DOM) pools

As mentioned, land-use conversions are often associated with an initial abrupt change and subsequent transition period following conversion. The 2006 IPCC Guidelines provide separate equations for these two phases when using Tier 2 and 3 approaches. Where country-specific emission factors are available and comprehensive national data are available, country-defined Tier 3 methodologies based on either process models or detailed inventories, stratified by climate and management regime can be recommended. These methods can also use non-linear loss and accumulation response curves during the transition phase.

At the other extreme, Tier 1 methods assume that both biomass and DOM pools are lost immediately after conversion from forestland to agricultural land, and that agricultural land reaches its steady-state equilibrium in the first year following conversion. While the IPCC provides default values to quantify biomass levels prior to and after conversion, there is assumed to be no accumulation in the DOM pool in the transition phase on agricultural land following conversion from forestland.

The Tier 2 methods represent a compromise, better capturing the dynamics of land-use conversion, by specifying separate equations for the abrupt change and transition phases, accounting for biomass accumulation during the latter phase. They also rely on some country-specific estimates of initial and final biomass stocks, instead of relying solely on default values.

Further, both Tier 2 and Tier 3 methods account for transfers between carbon pools and can estimate carbon pool changes using either the gain-loss or stock-difference methods. The former method includes all processes that cause changes in a carbon pool, including biomass growth and the transfer of carbon from one pool to another. Alternatively, the stock-difference method can be used where carbon stocks are measured at two points in time. Both methods are valid, providing they can represent disturbances and continuously varying trends, and can be verified with actual measurements (IPCC, 2006).

2.1.2 Soil organic carbon (SOC) stocks

Changes in the SOC pools in both mineral and organic soils should be taken into account when estimating emissions and carbon accumulation resulting from LUC (IPCC, 2006). This requires that the areas of converted land be stratified by climate region, management and major soil type. Simple Tier 1 methods, which rely on default reference SOC stock change factors, can be used, or more country- or region-specific reference C stocks and stock change factors can be combined with more disaggregated land use activity data to use either Tier 2 or Tier 3 methods. Some of the process models suited to Tier 3 methods are discussed in the following section.

In this study, LUC emissions are estimated for each major carbon pool, including the biomass, DOM and SOC pools are estimated using Tier 1 methods. While Tier 2 and Tier 3 methods are recommended, the Tier 1 approach was deemed to be appropriate given the global nature of the assessment combined with the absence of country-specific emission factors, inventory data and/or a suitable global process model.

2.2 Changes in carbon stocks for agricultural land remaining in the same land use category

As with LUCs, the estimation of emissions and carbon accumulation from management practices on land that remains in the same land use category requires that changes in carbon stocks from each major carbon pool (i.e., above-ground biomass, below-ground biomass, dead wood, litter and soil organic matter), as well as emissions of non-CO_2 gases, are estimated.

For agricultural lands, changes in these carbon pools and non-CO_2 emission fluxes depend on management practices such as grazing, burning, pasture management, tillage and residue management. Tier 2 and Tier 3 methods are able to estimate changes in each carbon pool and in emissions resulting from management practices, while Tier 1 methods can only be used to estimate these changes for the SOC pool (and non-CO_2 emissions from burning), but not for the other carbon pools. As with the measurement of emissions and carbon storage under LUC, the same gain-loss and/or stock-difference methods can be employed.

As discussed, Tier 3 methods can be used to more accurately assess changes in these carbon pools and non-CO_2 emission sources, using dynamic process models and/or detailed inventory measurements to estimate carbon stock changes. Process model-based approaches simultaneously solve multiple equations to estimate net changes in carbon stocks. These models can incorporate management effects such as grazing intensity, fire, fertilization, tillage and residue management, and they can be combined with regionally representative sampling-based estimates to validate and extrapolate to other agricultural lands. According to IPCC (2006), important criteria for selecting these models include: their ability to represent all relevant management practices and production systems, the compatibility of model's driving variables (inputs) with available country data, and validity gauged by the model's ability to represent stock change dynamics reported in empirical assessments. Well-known biogeochemical models that can satisfy these criteria include the Century model (and the daily time-step version, Daycent), DNDC and RothC.

The RothC (Hart, 1984; Jenkinson *et al.*, 1987; Coleman *et al.*, 1997; Smith *et al.*, 2006) and Century (Parton *et al.*, 1987; Falloon and Smith, 2002; Kirschbaum and Paul, 2002) models can be used to simulate GHG gas exchange and carbon cycling dynamics of cropland, grassland and forestland land use categories, and both operate on monthly time-steps. Soil texture and weather data are the major input variables. While the Century model can simulate the dynamics of carbon in biomass, DOM and SOC pools, as well as nitrogen, phosphorous, and sulphur dynamics, RothC only estimates SOC stocks and CO_2 losses from decomposition of SOC.

The Daycent model is the daily time-step version of the Century model (Del Grosso *et al.*, 2001; Parton *et al.*, 1998), which is well suited to capturing N mineralization and N gas production in non-waterlogged soils, along with the same carbon pool dynamics modelled in Century. As with Daycent, the denitrification-decomposition (DNDC) model (Li, 1996; Li *et al.*, 1992 and 1994) simulates soil carbon and nitrogen fluxes using a daily time-step but, unlike Daycent, it is also able to represent N gas and CH_4 fluxes from waterlogged soils, such as found in rice paddies. Both Daycent and DNDC have higher data demands than either Century or RothC, due their short time-steps and wider range of biogeochemical dynamics. Since none of these models has been validated on a global scale, they have not been applied in this analysis.

3. QUANTIFICATION OF CARBON STOCK CHANGES FROM LAND USE AND LAND-USE CHANGE IN THIS REPORT

In this study, LUC emissions are estimated for three major carbon pools, including the biomass, DOM and SOC pools. It could be argued that Tier 2 and Tier 3 methods, including process-based modelling approaches, should have been used to capture variability and possibly to reduce uncertainty in the emission and carbon accumulation estimates. However, given the global nature of the assessment, and the absence of country-specific EFs, carbon stock/flux inventory data and/or a suitable global process model (cf. previous section), the Tier 1 approach was deemed a suitable option to develop preliminary estimates and shed light on the potential magnitude of the LUC emissions for the sector.

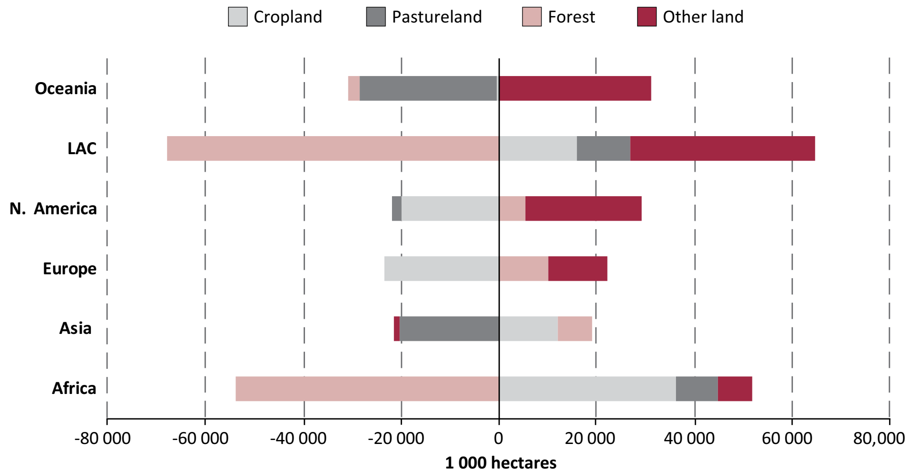
For the reasons outlined above, this assessment does not cover changes in C stocks occurring under constant land use management. This may be done in future updates once global datasets are available and/or models have been calibrated for global studies.

This section presents the approach applied in this study to quantify LUC emissions, discussing the rationale for the approach chosen, and the results from the analysis. It also explores the implications of alternative approaches to quantifying LUC emission.

3.1 Approach for feed crops

The analysis focuses on one specific feed product – soybean – in specific countries in Latin America. This assessment is based on observed land use trends, feed crop expansion trends and trade flow patterns as well as findings from previous studies such as Wassenaar *et al.* (2007) and Cederberg *et al.* (2011).

Figure C1.
Net land conversion between 1990 and 2006, by region



Source: FAOSTAT (2012).

This study uses IPCC guidelines as a basis for the quantification of LUC emissions. This choice is largely based on the fact that the IPCC approach meets the UNFCCC requirements for calculating and reporting of GHG emissions from LUC. The cropland part of this assessment also relies on other guidelines such as the PAS 2050 (also based on IPCC guidelines) for input data. According to IPCC guidelines, emissions arising from LUC are allocated over a 20-year period (the "amortization" period). Because of data availability (forestry inventories are only available from 1990),[15] in this assessment, the rates of LUC are taken as the average over the 16-year period (1990–2006). This practically discounts four years of emissions.

Agriculture has been a major driving force behind land transformation; globally, the area of land used for agriculture increased by 83 million ha over the period 1990–2006. In most regions, cropland has increased whereas pasture and forest land decreased (Figure C1). The most affected regions in terms of crop expansion are Latin America, Asia and Africa. Declining agricultural land (i.e. cropland and pastureland) is observable in Europe and North America where agricultural land abandonment has resulted in reforestation. During the period considered (1990-2006), deforestation occurred mainly in Africa and Latin America. More recent trends in deforestation, in particular in Asia, and their association with feed production are therefore not considered in this study.

Between 1990 and 2006, crop expansion was mainly driven by major oil crops (e.g. soybeans, rapeseed, sunflower and oil palm) the demand for which was fuelled by demand for vegetable oil, feed and, more recently, biofuel policies. The expansion of soybean production is argued to be one of the major drivers of LUC, par-

[15] The FAOSTAT forest area dataset (based on the Global Forest Resource Assessment) used in this study is only available from 1990 and in order to align the C stocks assessment with the livestock input data which is based on 2005 statistics, land use conversion trends were assessed for the period 1990 to 2006.

Table C1. Global area expansion for selected crops with highest area expansion (1990-2006)

Crop	Area expansion *(1 000 ha)*	Share of global gross crop expansion *(percentage)*
Soybeans	38 110	22.6
Maize	15 620	9.2
Rapeseed	9 815	5.8
Rice, paddy	8 650	5.1
Sunflower seed	7 237	4.3
Oil palm fruit	7 205	4.3

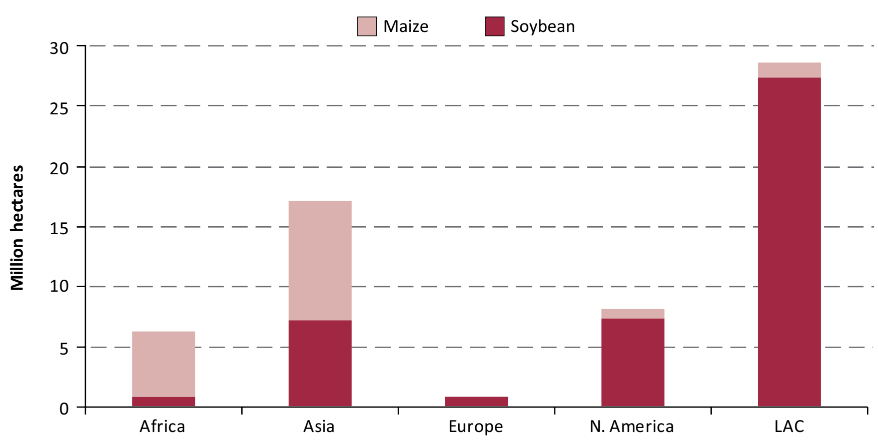
Source: FAOSTAT (2012).

ticularly deforestation (Pacheco, 2012; Nepstad *et al.*, 2006; Fearnside, 2005; Bickel and Dros, 2003; Carvalho *et al.*, 2002). The global area under cultivation of soybean has increased rapidly in recent decades; between 1990 and 2006, the global soybean area increased faster than any other crop (Table C1). Maize expansion is also important, representing 9.2 percent of global crop expansion. At the same time, crops such as wheat, barley and oats, have strongly declined, which explains the apparent discrepancies with the net land conversion trends in Figure C1.

A comparison of the two major crops driving agricultural expansion reveals key regional differences with regard to their importance (Figure C2). The expansion of soybean area has been significant in North and South America, while maize expansion is more important in Africa and Asia.

Deforestation for crop expansion has been an important LUC process in Africa, however crop expansion in the region has been mainly driven by sorghum and millet, with maize and soybeans only accounting for 5 percent and 0.5 percent of total gross cropland expansion respectively. In Africa, pasture expansion has also occurred largely at the expense of forest area. However, due to lack of reliable data and information it is difficult to draw conclusions on the land-use conversion trends in this region.

Figure C2.
Maize and soybean area expansion between 1990 and 2006, by region



Source: FAOSTAT (2012).

Table C2. Average annual land-use change rates in Argentina and Brazil (1990-2006)

Land-use type	Argentina	Brazil
	(1 000 ha)	
Agricultural area	+351	+1 288
Grasslands	-7	+753
Arable land & permanent crops	+358	+535
Soybean area	+648	+534
Forest area	-149	-2 855
Other land	-201	+1 567

Source: FAOSTAT (2009).

In North America, soybean expansion is responsible for 37 percent of total crop expansion and maize 7 percent. However in this region the overall trend has been a decrease of total cropland (due to sharp decreases in wheat and barley areas) and pastures and an increase of forest area.

In Asia, soybean expansion is responsible for 7 percent of total crop expansion and maize 8 percent. At the same time, forest land has increased overall in Asia and pastureland has decreased. But the two trends occurred in different subregions within Asia. Pasture decrease mainly occurred in Mongolia and Iran, where maize and soybean expansion were null or limited. On the contrary, expansion of soybean and maize area has largely occurred in India and China (77 percent of gross maize expansion and 96 percent of gross soybean expansion), however, forest area increased in these two countries. Pastures decreased in India but to a limited extent of 1.2 million ha, compared to the 5.8 and 3.0 million ha of soybean and maize expansion in the country, respectively.

In Latin America, most of the decrease in forest area occurred in countries with soybean expansion. Trends in land conversion, particularly deforestation, are therefore closely linked to the expansion of soybean.

Based on these observations the scope of our assessment was narrowed to the soybean expansion in Latin America. Within Latin America, Brazil and Argentina account for 91 percent of the total soybean area. In the period 1990–2006, 90 percent of the soybean area expansion in Latin America took place there, further narrowing the scope to these two countries. An assessment of land use trends in these key producing countries shows that the expansion in soybean area has been largely gained at the expense of forest area (Table C2).

In Argentina, the annual increase of area dedicated to soybean is much larger than the increase of total arable land (Table C2), indicating that there has been a shift in land use from other crops to soybean. According to FAOSTAT statistics, 44 percent of the new soybean area was gained against other crops, while the rest was gained against forest (22 percent) and other land (31 percent). The latter category covers natural vegetation that does not include forest and grazed natural grasslands.

The reported annual increase of soybean area in Brazil is 534 000 ha (Table C2). We assumed a simplified pattern of deforestation in the Amazon, in which cleared land is first used as pasture and/or crop land, and then left as fallow land. The latter, classified as "other land" in FAOSTAT, is occupied by weeds, grasses, shrubs and,

Table C3. Net changes in area for main land-use categories (1990–2006)

Region	Arable land & permanent crops	Pasture	Forest area	Other land
		(1 000 ha)		
Africa	36 025	8 863	-53 700	7 001
Asia* (*South, East and SE Asia*)	12 149	-20 506	6 855	-1 068
Europe	-55 646	-152 441	261	-96 796
North America	-20 073	-1 954	5 387	23 811
Latin America and the Caribbean	15 753	11 069	-67 870	37 973
Oceania	-263	-28 408	-2 112	30 926

* Central Asia excluded due to incomplete dataset.
Source: FAOSTAT (2012).

partly, by secondary forest. Under this assumption, every year roughly 2.9 million ha are converted to arable land and grassland. At the same time, agricultural land is abandoned at a rate of 1.6 million ha per year. The annual net increase of arable land and grassland is 0.53 and 0.75 million ha, respectively. We thus assume that all incremental soybean area is gained at the expense of forest area.

Rates of C loss/gain arising from specific land-use transitions were taken from PAS 2050 guidelines (BSI, 2008), which are based on IPCC (2006). The PAS 2050 guidelines estimate deforestation (conversion of forest to annual cropland) releases in Brazil at an average 37 tonnes CO_2-eq/ha, and conversion of forest and shrub land to annual crop in Argentina at 17 and 2.2 tonnes CO_2-eq per ha, respectively. GHG emissions from soybean-driven LUC were calculated as the accumulated emissions for one year resulting from the total area deforested during the period 1990–2006 divided by the total soybean production in 2006. Based on this data, two LUC emission intensities were estimated for soybean cake produced in Brazil and Argentina, respectively: **7.69 and 0.93 kg CO_2-eq/kg soybean cake**. Soybeans and soybean cake produced elsewhere were assumed not to be associated with LUC.

3.2 Pasture expansion and land-use change

It has been argued that while forest conversion to soybean cultivation is occurring, the majority of deforested area is destined to pasture formation (Morton *et al.*, 2006; Brown *et al.*, 2005). Wassenaar *et al.* (2007) developed a spatial and temporal model framework to analyse the expansion of pasture into forest in Latin America. The analysis predicted that, on average, 76 percent of deforested land would become pasture. Table C3 presents the net changes for different land use categories across regions; pasture expansion has been notable in Latin America and Africa while, at the same time, forest area in Latin America and Africa during the same period declined substantially.

3.2.1 Approach

The approach is based on the IPCC stock-based approach termed the *Stock-Difference Method*, which can be applied where carbon stocks are measured at two points in time to assess carbon stock changes (IPCC, 2006). The following emissions from deforestation were considered:

- CO_2 emissions from changes in biomass stocks (above-and below-ground biomass);
- CO_2 emissions from changes in dead organic matter (litter and deadwood);
- CO_2 emissions from changes in soil carbon stocks.

For each of the carbon pools mentioned above, several factors such as land use (forest, croplands, pasture), climatic zone, ecotype (tropical moist or tropical dry forest), soil type (mineral or organic soils), forest type, etc., were taken into consideration. Since data from forestry inventories are only available from 1990, the changes in carbon stocks due to deforestation could only be calculated for the period 1990-2006.

The calculations of land-use change were accomplished in two steps: first, the assessment of land use dynamics; and second, the carbon emissions based on land use dynamics and biophysical conditions. A complete assessment of carbon emissions from LUC involves the quantification of several key elements including deforestation rates, land use dynamics, and initial carbon stocks in biomass and soil. Two types of information are fundamental to enable emissions to be calculated: rates of deforestation and per hectare changes in carbon stocks in the different carbon pools. The following sections provide a detailed description of the applied methodology and assumptions made.

Determining total land area converted from forest to grassland. To accurately estimate carbon fluxes from LUC, it is critical to understand LUC dynamics following deforestation. With regard to land-use transition matrices, a simplified approach was adopted. Changes in land use area were estimated on the basis of the Tier 1 approach outlined in Chapter 3 of the IPCC guidelines, which estimates the total change in area for each individual land use category in each country.

FAOSTAT statistics on total land area (classified by land use category) were used to calculate the annual net change in the area of each land use category.

Table C4 presents the countries in which the increase in pasture area was largely facilitated by a decrease in forest area, and our estimates show that about 13 million hectares were deforested for pasture establishment.

Table C4. Pasture expansion against forestland in Latin America (1990-2006)

Countries	Pasture area change *(1 000 ha)*	Share of regional expansion *(percentage)*
Brazil	10 212.3	77.2
Chile	1 150.0	8.7
Paraguay	1 040.0	7.9
Nicaragua	454.3	3.4
Other*	365.0	2.8
Total	**13 221.6**	**100**

* 'Other' category includes: Honduras, Ecuador, Panama, El Salvador and Belize.
Source: Authors' calculations based on FAOSTAT data.

Table C5. Country specific estimates of above-and below-ground biomass

Countries	Above-ground biomass[1] (tonnes DM/ha)	Ratio of below- to above- ground biomass[2]	Below-ground biomass (tonnes DM/ha)	Total biomass[3] (tonnes DM/ha)
Brazil	220	0.24	52.8	272.8
Chile	220	0.24	52.8	272.8
Paraguay	210	0.24	50.4	260.4
Nicaragua	210	0.24	50.4	260.4
Ecuador	300	0.37	111	411.0
Other*	220	0.24	52.8	272.8

[1] Derived from IPCC, Volume 4, Chapter 4, Table 4.4.
[2] Ratio of above-below ground factors are derived from IPCC guidelines, Table 4.7.
[3] Total biomass is the sum of above-and below-ground biomass.
* 'Other' category includes: Honduras, Ecuador, Panama, El Salvador and Belize.

Changes in carbon stocks from above- and below-ground biomass. The method applied here focuses on stock changes in biomass associated with woody vegetation which are capable of accumulating large quantities of carbon over a long period of time. The Tier 1 method necessitates the estimation of biomass before and after conversion using IPCC equation 2.16 (Volume 4, Chapter 2).

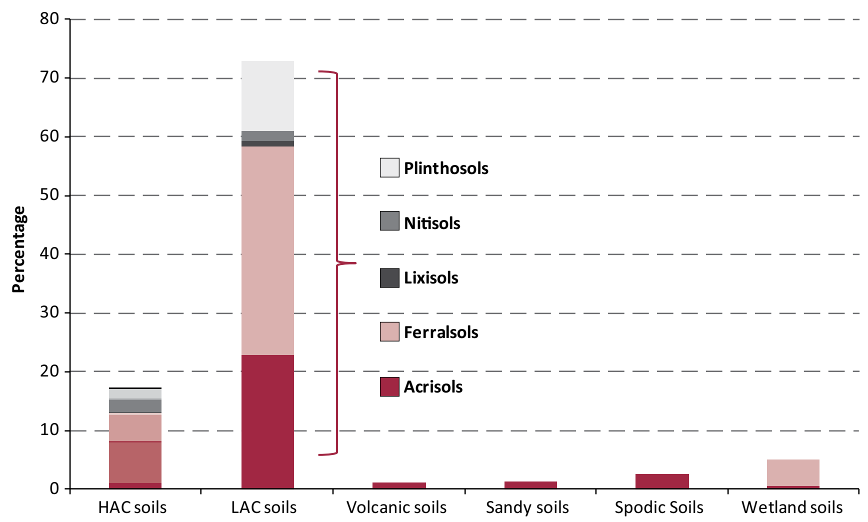
Biomass in forests is determined by ecological zone, type of native vegetation and geographical location of forests. Based on the IPCC Tier 1 approach, in the conversion of forest to grassland it is assumed that all biomass is cleared and therefore the default biomass after conversion is 0 tonnes DM ha^{-1}. The IPCC guidelines (Chapter 4, Volume 4, Table 4.7) provide average default values for above-ground biomass in forests. Due to the lack of data on below-ground biomass, the ratio of below-to-above ground biomass (root-to-shoot ratio) was used to estimate the below-ground component of biomass and the total biomass (tonnes DM/ha) given in Table C5. A default factor of 0.50 tonnes C per tonnes DM (carbon fraction for woody biomass was used to convert biomass into carbon stocks per hectare.

Changes in carbon stocks from dead organic matter (DOM) pools. The conceptual approach to estimating changes in C stocks in dead wood and litter pools is to estimate the C stocks in the old and new land use categories and apply this change in the year of conversion. Equation 2.23 (IPCC, 2006, Volume 4, Chapter 2) was used to estimate changes in C stocks from DOM.

According to the IPCC Tier 1 approach, DOM pools in non-forest land categories after the conversion are zero and this is based on the assumption that all DOM carbon losses occur entirely in the year of land-use conversion. Tier 1 also assumes that carbon contained in biomass killed during the conversion of land is emitted to the atmosphere and none is added to the dead wood and litter pools. Tier 1 default factors for dead wood and litter were taken from IPCC (2006, Volume 4, Chapter 2, Table 2.2).

Changes in soil carbon stocks. SOC stock changes do not occur instantaneously but over a period of years to decades. The current IPCC good practice guidance for GHG inventories assumes a period of 20 years for a new equilibrium to occur after conversion (IPCC, 2006). The change in the amount of SOC depends on factors such as climate region, native soil type, management system after conversion

Figure C3.
Main soil classes in Latin American forested areas based on the World Reference Base for Soil Resources classification



LAC: low activity clay; HAC: high activity clay.
Source: Authors.

and input use. The calculation of SOC losses per hectare of area transformed from forest to grassland is based on equation 2.25 in IPCC (2006, Volume 4, Chapter 2), which takes into account changes in soil carbon stocks associated with type of land use, management practices and input of organic matter (fertilization, irrigation, liming and grazing intensity) in the soil.

The approach makes a distinction between organic and mineral soil carbon pools, and the focus is on the impacts of LUC on the organic pool, because inorganic soil carbon is assumed to be insensitive to land-use change and management. Land converted to grassland was stratified according to climatic region, management and major soil types based on country specific classifications. The starting point was to derive a soil type classification of areas under forest in the selected countries in order to determine SOCs. This was accomplished with overlays of suitable climatic and soil maps coupled with spatial data on forest land area[16,17]. Figure C3 presents the mapping results of country-specific soil types on forested land and provides information on the dominant soil groups in forested areas in Latin America.

To establish SOC stocks, the soil divisions were further aggregated into dominant soil type classes (Figure C3) defined in IPCC guidelines based on the World Reference Base (WRB) for Soil Resources classification. Based on this aggregation, at a regional level, soils with low activity clay cover nearly 73 percent of the forested area in the nine countries. The remaining forested area is made up of the other five dominant soil types of which the high activity clay soil types cover 17 percent of the area (Figure C3).

[16] FAO/IIASA/ISRIC/ISS-CAS/JRC. 2009. Harmonized World Soil Database (version 1.1). FAO, Rome, Italy and IIASA, Laxenburg, Austria.

[17] Arino, O., Ramos, J., Kalogirou, V., Defourny, P. & Achard, F. 2010. "GlobCover 2009", Proceedings of the Living Planet Symposium, SP-686, June 2010. Data downloaded from http://ionia1.esrin.esa.int/ in August 2011.

Table C6. Default soil organic C stocks for mineral soils

Climate region	High activity clay soils	Low activity clay soils
	(tonnes C.ha^{-1} in 0-30 cm depth)	
Boreal	68	NA
Cold temperate, dry	50	33
Cold temperate, moist	95	85
Warm temperate, dry	38	24
Warm temperate, moist	88	63
Tropical, dry	38	35
Tropical, moist	65	47
Tropical, wet	44	60
Tropical, montane	88	63

NA: Not Applicable because these soils do not normally occur in some climatic zones.
Source: IPCC (2006).

Table C7. Soil organic carbon pool at 0-30 cm depth

Countries	Soil C stocks under forest	Soil C stocks under grassland	Net change in carbon stocks	Net annual change
	tonnes C.ha^{-1}		*tonnes C.ha^{-1}*	*tonnes C.ha^{-1} yr^{-1}*
Brazil	60	58.20	-1.8	- 0.11
Chile	44	42.68	-1.3	- 0.08
Paraguay	65	63.05	-2.0	- 0.12
Nicaragua	35	33.95	-1.1	- 0.07
Honduras	56	54.32	-1.7	- 0.11
Ecuador	78	75.66	-2.3	- 0.15
Panama	65	63.05	-2.0	- 0.12
El Salvador	50	48.50	-1.5	- 0.09
Belize	65	63.05	-2.0	- 0.12

Source: Authors' calculations based on IPCC (2006).

The 2006 IPCC guidelines provide average default SOC stocks for the dominant soil classes clustered by eco-region reproduced in Table C6. The default reference soil organic C stocks are for the top 30 cm of the soil profile because different land use management methods mostly affect soil carbon in the surface layer.

For Tier 1, all stock change factors (F_{lu}, F_{mg}, F_I) were assumed to be equal to 1 for forest land, corresponding to the default values in IPCC guidelines. For grasslands, stock change factors used for land use and input (G_{lu}, *and* F_I) were assigned a value of 1.

The quality of management of tropical pastures after conversion is critical in understanding whether the soils under this land use represent a source or a sink of atmospheric carbon. Differences in management practices could significantly affect subsequent trends in soil carbon. Due to the limited data on management and input, default values were used.

It was assumed that pastures are moderately degraded and therefore a coefficient of 0.97 (IPCC, 2006 Volume 4 , Chapter 6, Table 6.2) for F_{mg} stock factor was applied, which represents overgrazed or moderately degraded grasslands with reduced productivity and receiving no management inputs. This assumption is based on the findings of studies (Hernandez *et al*., 1995; Murty *et al*., 2002; De Oliveira *et al*., 2006; Cerri *et al*., 2005;) which inferred that most of the pastures in LAC are in some process of degradation caused by poor management methods, low input fertilization and no maintenance. The results (Table C7) show a net decrease in SOC with losses ranging between 1.1 to 2.3 tonnes C ha^{-1}.

3.3 Sensitivity analysis and the influence of LUC method

Modelling of land use and LUC emissions is subject to great uncertainties mainly because of the complexity of LULUCF processes, the challenges of obtaining reliable global data and the absence of validated approaches to estimate carbon stock changes. In particular, uncertainty regarding the magnitude of LUC emissions arises due to uncertainties in: (a) the rates of land use; (b) the carbon storage capacity of different forests, initial carbon stocks and the modes of C release; and (c) the dynamics of land use not normally tracked. In addition, a value judgment has to be made regarding what drives LUC and, consequently, how the emissions should be allocated. In order to explore the potential effect that different methodologies can have, the results obtained with the GLEAM approach are compared to three alternative approaches: (a) PAS 2050-1:2012; (b) One-Soy; and (c) reduced time-frame approach. These approaches are summarized in Table C8.

3.3.1 Alternative approaches

PAS 2050-1: 2012 approach. Several studies suggest that deforestation is related to the expanding soybean sector (Fearnside, 2005; Bickel and Dros 2003; Carvalho *et al.*, 2002), but others dispute this claim, and argue that soybean is expanding into land previously under pasture, and is not causing new deforestation (Mueller, 2003; Brandao *et al.*, 2005). Due to the lack of knowledge of the origin of the converted

Table C8. Alternative approaches for soybean LUC emissions calculations

Method	Spatial allocation	Temporal allocation of LUC emissions	Quantification of rates of LUC	Quantification of rates of C loss/gain
GLEAM approach (current study)	To all soybean produced within the country	20 years	FAOSTAT average LUC rates 1990-2006 Brazil: forest→crops (100%) Argentina: other crops (44%), forest (22%) and other land (31%)→soybean	IPCC (2006) Tier 1
PAS 2050-1:2012	To all soybean produced within the country	20 years	Average rates over 20 years. LUC rates based on (a) or (b) - whichever results in the highest emission factor. (a) from grassland forest and perennial arable in equal proportion (b) from grassland, forest and perennial arable in proportion to their rates of change	IPCC (2006) Tier 1
One-Soy	To traded soybean	20 years	FAOSTAT average LUC rates 1990-2006 Brazil: forest→crops (100%) Argentina: other crops (44%), forest (22%) and other land (31%)→soybean	IPCC (2006) Tier 1
Reduced time-frame	To all soybean produced within the country	20 years	FAOSTAT average LUC rates 2002-2007 Brazil: forest→crops Argentina: other crops (44%), forest (22%) and other land (31%)→soybean	IPCC (2006) Tier 1

Source: Authors.

land, the GLEAM results were compared with PAS 2050-1:2012 (BSI, 2012), which provides a way of quantifying LUC emissions when previous land use is not known and only the crop and country are known. The PAS 2050-1:2012 calculations of emissions related to land-use change are accomplished in two steps.

First, rates of land-use change need to be calculated based on the PAS 2050-1: 2012. To calculate these, four categories of land are considered: forest, pasture, annual cropland and perennial cropland. Time series data on land area for forest, pasture, annual and perennial crops taken from FAOSTAT were used to: (i) determine whether the crop in question was associated with LUC by quantifying the rate of expansion over a 20-year period; and (ii) determine the share of LUC associated with each land category. In a second step, carbon losses based on land dynamics and biophysical conditions (climate, soil type, forest type, crop management, etc.) were computed based on the IPCC (2006) Tier 1 approach. The two sources of carbon taken into account in this approach are vegetation and soil. Two LUC EFs were calculated, based on different assumptions regarding where land for soybean expansion is derived from: (i) assuming that land for soybean production is gained in equal proportions from grassland, forest and perennial cropland; (ii) assuming that land for soybean is gained from other land use categories in proportion to their relative rates of change. The highest of the two EF's was then selected, in accordance with the guidelines. BSI (2012) present a detailed account of methodology and data sources.

One-Soy approach. In this approach it is assumed that all soybeans, irrespective of where they have been produced, are associated with LUC. The central argument for this scenario is that the global demand for soybeans is largely interconnected and is a key driver of LUC. An average LUC emission factor associated with soybean was estimated by calculating the total LUC emissions attributable to globally-traded soybean and soybean cake and then dividing this by total global soybean cake exports. Because the emission intensity was applied to all traded soybean and soybean cake, the approach equally distributes the LUC emissions across all importing countries irrespective of where the soybean is produced.

Reduced time-frame approach. Annual deforestation rates are highly variable, so the period over which the rates of LUC are estimated can therefore have a significant influence on results. Since data from forestry inventories are only available from 1990, this assessment was based on the average rates of LUC over the period 1990-2006. This not only coincides with a period of high rates of deforestation but also high soybean area expansion. In the reduced time frame approach, the LUC emissions are calculated based on the average rates of LUC over the period from 2002-07, while maintaining the underlying assumptions in the study.

3.3.2 Results

Effect of LUC approach on soybean LUC emission factor. Table C9 reports the LUC factors for soybean cake (kg CO_2-eq per kg soybean cake) calculated using each of the approaches. The choice of method for estimating LUC EFs can strongly influence the emission intensity of livestock products and illustrates the complexity of analysing LUC processes.

Table C9. Summary of soybean LUC emission intensity for the four approaches

Scenario	Argentina	Brazil
	(kg CO_2-eq per kg soybean cake)	
GLEAM approach (current study)	0.93	7.69
PAS 2050-1:2012	4.23	3.21
One-Soy	2.98	2.98
Reduced time-frame	0.34	3.70

Source: Authors' calculations.

The *PAS 2050-1: 2012* approach produces markedly different LUC emission factors due to the assumptions made regarding the land use category against which additional land for soybean production was gained and the relative share of this gain (Table C10). Unlike Brazil, Argentina has a higher EF using the default assumption (that expanded crop areas are derived from forest, grassland and perennial crops in equal proportion) than using the relative rates of change. The higher proportion of soybean cultivated on expanded areas in Argentina (76 percent) compared to Brazil (55 percent), combines with the default LUC assumptions, to give Argentina a higher soybean EF than Brazil under PAS 2050-1:2012.

The strength of the *One-Soy* approach is that it recognizes that global demand is a key driver of LUC. However, it penalizes those countries whose production is not directly associated with LUC and may not provide the right signals to producers and consumers of soybean.

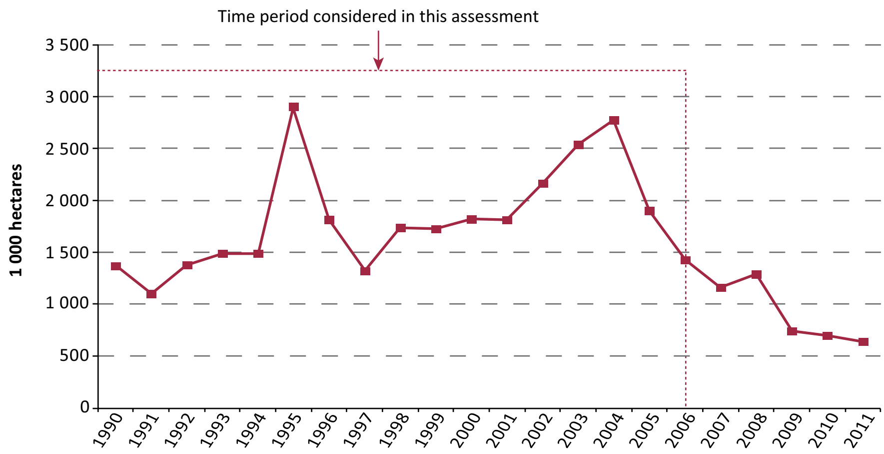
In the *reduced time-frame approach*, the emission intensity of soybean cake from Argentina and Brazil reduces by more than half. Average annual deforestation rates appear to be close over the two periods 1990-2006 and 2002-2007 (1.76 and 1.98 million ha respectively, Figure C4), but the average annual rates of soybean expansion differ and they are higher for 2002-2007: between 1990 and 2006, the soybean area in Brazil increased by 534 000 ha/year whereas the increase for the period 2002-2007 was 840 000 ha/year. The lower emission intensity for 2002-2007 therefore results from the rate of deforestation relative to the rate of soybean expansion, not from the absolute change in deforestation rate.

Table C10. Proportion of expanded soybean area derived from each land-use category

Land-use category	GLEAM approach		PAS 2050-1:2012 approach	
	Brazil	Argentina	Brazil	Argentina
	percentage			
Forest	100	22	51 (33)	23 (33)
Grassland	0	0	0 (33)	0 (33)
Shrubland	0	31	0 (0)	0 (0)
Annual cropland	0	44	46 (0)	61 (0)
Perennial cropland	0	0	3 (33)	16 (33)

Note: Figures in brackets are the PAS 2050-1 default land use transformations.
Sources: Based on FAOSTAT (2012).

Figure C4.
Annual forest loss in Brazil



Source: INPE, 2012.

Table C11. Estimated changes in pasture area and annual carbon losses for the reduced time-frame approach

Country	Change in pasture area *(1 000 ha)*	Carbon losses *(tonnes CO_2/ha/year)*
Brazil	3 563.6	- 51.0
Chile	1 079.3	- 51.1
Paraguay	1 464.6	- 48.8
Nicaragua	311.6	- 48.5
Honduras	170.0	- 50.8
Ecuador	No gain in pasture	-
Panama	60.3	- 50.8
El Salvador	45.3	- 50.8
Belize	0.3	- 51.1
Total/average	**6 695.3**	**- 50.4**

Source: Authors' calculations.

For pasture expansion, emissions are highly sensitive to the time period chosen; using a ten-year time-frame scenario, annual carbon losses are 50.4 tonnes CO_2 ha^{-1} yr^{-1} (Table C11) while in the current study annual carbon losses were estimated at 32 tonnes CO_2 ha^{-1} yr^{-1}. Shorter periods, however, place emphasis on deforestation resulting in higher annual carbon losses per hectare placing higher relative weighting of near-term emissions.

4. COMPARISON WITH OTHER STUDIES

The emissions intensity for LUC per kg of soybean and soybean cake calculated in this study are compared with other studies in Table C12. The emissions intensity used in this study is higher than some other studies, but within the overall range.

The emissions intensity of soybean is highly dependent on the calculation method and assumptions (Flysjo *et al.*, 2012). Variation arises from differences in:

- Calculation of C losses in soil and vegetation (above- and below-ground).
- Quantification of land-use transitions – i.e. how much of the LUC can be attributed to cropping.
- Allocation of LUC arising from cropping to specific crops, e.g. emissions are usually allocated to one of the following: (a) soybean grown in country/region; (b) all expanding crops grown in country region; (c) all crops grown globally. This leads to huge variations in the emissions per kg of crop.
- The time period over which emission are allocated.

Table C12. Soybean LUC emissions per unit of output and hectare

Study	Area covered by study	Emissions	*Converted/all soybean /all crops
FAO (2010)	Argentina	1.04 kg CO_2-eq/kg soybean	All soybean
FAO (2010)	Brazil	7.69 kg CO_2-eq/kg soybean cake	All soybean
FAO (2010)	Brazil	8.54 kg CO_2-eq/kg soybean cake	All soybean
FAO (2010)	Brazil	12.81 kg CO_2-eq/kg soybean cake	Converted
FAO (2010)	Brazil	14.23 kg CO_2-eq/kg soybean	Converted
Leip *et al.* (2010) grass>soybean	South America	1.50 kg CO_2-eq/kg soybean cake	All soybean Cited in Flysjo *et al.* (2012)
Leip *et al.* (2010) mix>soybean	South America	3.10 kg CO_2-eq/kg soybean cake	All soybean Cited in Flysjo *et al.* (2012)
Leip *et al.* (2010) forest>soybean	South America	10.00 kg CO_2-eq/kg soybean cake	All soybean Cited in Flysjo *et al.* (2012)
Sonesson *et al.* (2009, p13)	Brazil	1.50 kg CO_2-eq/kg soybean	All soybean ~0.6 of this is due to LUC
Audsley *et al.* (2010, p.59)	Brazil	5.30 kg CO_2-eq/kg soybean	All soybean
Audsley *et al.* (2010, p.59)	Argentina	1.60 kg CO_2-eq/kg soybean	All soybean
Castanheira & Freire (2011)	Low (Argentina)	~0.5 kg CO_2-eq/kg soybean	Converted
Castanheira & Freire (2011)	High (Brazil)	~15 kg CO_2-eq/kg soybean	Converted
Nemecek *et al.* (2012)	Brazil	1.47 kg CO_2-eq/kg soybean	All soybean Brazil, LUC, Ecoinvent v2.2
Nemecek *et al.* (2012)	Brazil	5.21 kg CO_2-eq/kg soybean	All soybean Brazil, LUC, Ecoinvent v3.0
Reijnders & Huijbregts (2008)	Brazil – cerrado	1 to 2.7 kg CO_2-eq/kg soybean	Converted
Reijnders & Huijbregts (2008)	Brazil – forest	5 to 13.9 kg CO_2-eq/kg soybean	Converted
FAO (2010)	Brazil – deforestation	37.00 kg CO_2-eq/ha	Converted
FAO (2010)	Brazil – deforestation	22.20 kg CO_2-eq/ha	All soybean
Audsley *et al.* (2009)	All LUC	1.43 kg CO_2-eq/ha	Allocates LUC to all crops globally
Audsley *et al.* (2010, p.59)	Brazil – deforestation	37.00 kg CO_2-eq/ha	Converted
Audsley *et al.* (2010, p.59)	Brazil – grassland	11.00 kg CO_2-eq/ha	Converted
Reijnders & Huijbregts (2008)	Brazil – forest	14 to 39 kg CO_2-eq/ha	Converted
Schmidt *et al.* (2011)	All LUC	8.42 kg CO_2-eq/ha	Allocates LUC to all crops globally

**EF* for (a) converted land; (b) average over all soybean grown in country/region; or (c) all crops grown globally.

Table C13. Comparison of studies on LUC associated with pasture expansion in Brazil

Study and area	Approach	Scope	Carbon losses *(tonnes CO_2-eq /ha)*
Current study (Brazil)	IPCC stock-based approach (stock difference method) Period: 1990-2006	Biomass Soil carbon Dead organic matter	506.7
Cederberg *et al.*, 2011 (Brazil -Legal Amazon Area)	Net committed emissions approach Period: 1986-2006	Biomass Soil carbon CH_4 and N_2O	572
Leip *et al.*, 2010 (Beef imported into EU from Brazil)	Net committed emissions approach Period: 1986-2006	Biomass Soil carbon CH_4 and N_2O	568.7

For pasture expansion, with the exception of Brazil where impacts of deforestation have been analysed to a greater degree, there are relatively few estimates of the impact of carbon losses the due to deforestation. We therefore compared the results obtained for Brazil with estimates from other studies (Table C13).

Despite the difference in calculation approach, our estimates are very similar to those found in the literature. This may be partly coincidental because the approaches differed in many respects; for example, the period assessed, the calculation method and assumptions and well as emission factors. Cederberg *et al.* (2011) apply different carbon stock losses for the different pools and take into account the impacts on fire used in forest clearing on CO_2 emissions.

The estimates of LUC emissions presented in this report are still very preliminary and need to be interpreted with caution. This is an important area for improvement of GLEAM and it is planned that future developments of the model will include a more detailed and complete assessment of LUC emissions.

5. LAND USE

For the reasons explained above, this analysis could not incorporate C stock changes under constant land use. This section attempts to evaluate the effect of this simplification on results. Given the importance of grasslands as a potential as a C sink (Soussana *et al.*, 2010), we focus our case study on this land use rather than on feed-crops.

Furthermore, we selected the European Union for this evaluation in view of data availability in this region. National inventories in the European Union are indeed increasingly accurate because Member States are requested to maintain and monitor the area of permanent grassland by the Common Agricultural Policy. Member States are required to report annual estimates of their total area of permanent grassland.

Soussana *et al.* (2010) estimate an average grassland C sequestration rate of 5 ± 30 g C/m^2/year for temperate grasslands under baseline, constant land use. This estimate is derived from an exhaustive literature review, and inventories of SOC stocks at regional or local level, mainly from Western Europe.

Using this estimate, we computed that permanent grasslands in the European Union (estimated at 62.7 million ha) represent a sink of 3.1 ± 18.8 million tonnes C per year, equivalent to 11.5 ± 69.0 million tonnes CO_2-eq per year. This estimate

Table C14. Total GHG emissions from the ruminant sector in the European Union and changes in C stocks in permanent pasture

Systems	Total emissions based on a LCA approach and excluding land use[1]	C sequestration in permanent grassland under baseline constant management[2]
	(million tonnes CO_2-eq/year)	
Total ruminant sector	**390.5**	**11.5 ± 69.0**
Cattle, grazing	25.6	
Cattle, mixed	322.8	
Small ruminants, grazing	4.6	
Small ruminants, mixed	37.4	

[1] Based on GLEAM.
[2] Based on Soussana *et al.* (2010).

is compared with the 390.5 million tonnes CO_2-eq emitted yearly by the ruminant sector in the European Union (Table C14).

Taking into account C stock changes in permanent pastures, net emissions from the EU ruminant sector are therefore estimated between 310 and 448 million tonnes CO_2-eq/year.

Net sequestration/emission of C in permanent pasture under stable management practices may thus be significant in the European Union, and should be included in the assessment of GHG emissions of the sector. The estimate computed here is however one order of magnitude smaller than the sum of all other emissions along the supply chain. Furthermore, even in a region where data availability is comparatively high, the uncertainty about C fluxes is such that it cannot be ascertained if permanent grasslands are net sequesters or emitters of carbon.

The European Union only accounts for a limited share of total grassland area (about 2 percent according to FAOSTAT, 2013), so including land use sequestration/emissions could have even greater effects on net emissions of the sector in other regions. For example, Cerri *et al.* (2004) measured that brasilian pastures established in the early 90's could store up to 330 g.m^{-2} of carbon in the 20 first centimeters of soil. This would however require a better understanding SOC dynamics in grasslands and the development of models and databases to monitoring and predicting changes in C stocks.

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APPENDIX D

Postfarm emissions

GHG emissions accounted for in the post farmgate part of the supply chain include emissions related to fuel combustion and energy use in the transport, processing and refrigeration of products. The system boundary is from the farmgate up to the retail point. During this phase of the life cycle, three distinct emission streams were studied: emissions from the transport and distribution of live animals, milk and meat (domestic and international); GHG emissions from processing and refrigeration; and emissions related to the production of packaging material.

The system boundary for this part of the food chain included emissions from the farmgate to the retail distribution centre. Excluded from the analysis were estimates of GHG emissions from on-site waste-water treatment facilities, emissions from animal waste[18] at the slaughter site, the retail and consumption part of the food chain (household transport and preparation) and disposal of packaging and waste, which fall outside the scope of the system boundary studied but may warrant further research. Due to the lack of data, emissions related to by-products (rendering material, offal, etc.) are therefore currently excluded. However, we investigated the impact of allocating emissions to slaughter by-products (Appendix F).

1. APPROACH AND ASSUMPTIONS

1.1 Milk from ruminant species

The quantification of post farmgate emissions for milk produced by cattle, small ruminants and buffalo was based on a similar approach. The approach (and level of complexity) was largely influenced by two factors: the importance of the subsectors contribution to global milk production and the availability of data. Consequently, a more comprehensive approach was applied to the milk from the global cattle dairy sector, as outlined in FAO's report on GHG emissions in the dairy sector published in 2010.[19]

In the estimation of post farmgate emissions for small ruminant and buffalo milk, a similar but simplified approach was adopted; for small ruminants, it was assumed that all milk that left the farm was processed into cheese. FAOSTAT production statistics on goat and sheep cheese production were used to identify countries where cheese production is important. In producing countries with no cheese production, the sheep and goat milk was assumed to be consumed on farm and hence no post farmgate emissions were estimated for these countries. In addition, since not all milk is processed and traded, the proportion of small ruminant milk leaving the farm was estimated from the cheese production and total milk production within the country.

[18] In some countries, manure/slurry from the slaughterhouse is anaerobically digested and the biogas is used for heating and electricity. The challenge is that there insufficient information available on on-site energy generation from animal waste; thus, the resulting substituted energy and avoided GHG emissions are not considered in the calculations.

[19] FAO. 2010. http://www.fao.org/docrep/012/k7930e/k7930e00.pdf

The difference in calculation approach between cow's milk and buffalo milk is that for dairy cattle milk six products were considered (processed milk, cheese, whey, yoghurt, skimmed milk powder and whole milk powder), while post farmgate emissions for buffalo milk comprised emissions related to transport and processing of raw milk into processed milk. Emissions related to international trade of dairy products was only considered for the cattle dairy sector.

1.2 Meat from ruminant species

GHG emissions reported for this part of the food chain are based on the finished product leaving the facility and do not account for meat co-products and rendering products; however, in a life cycle assessment, when a system produces multiple products each of which have economic value, it is standard practice to assign some of the emissions from that process to each of the co-products.

In this analysis, all emissions were allocated to the carcass and therefore meat carries the whole burden. Post farmgate emissions for meat include: emissions associated with the transport of live animals to slaughterhouses, emissions related to slaughter and primary processing of carcasses, refrigeration of carcasses at processing plant and transport and refrigeration of product. Emissions related to international trade of meat products (carcasses and boneless meat) are taken into account. Due to the complexity of tracking trade flows of live animals the related emissions are excluded from this analysis.

2. ENERGY CONSUMPTION

Energy consumption is the most important source of GHG emissions from the post farmgate supply food chain. Table D1 presents average regional and country CO_2 emission coefficients applied in this analysis. The CO_2 intensities are determined by the composition of the energy sources employed and average GHG emissions from electricity consumption was modelled as a mix of existing electricity sources (e.g. coal, hydro, nuclear, oil, etc.) in different countries and regions taken from the International Energy Agency (IEA, 2009).

Table D1. Average regional specific CO_2 emissions per MJ from electricity and heat generation

Region/country	CO_2 emissions *(g CO_2/MJ)*
Europe 27	99
North America	142
Australia	254
New Zealand	84
Japan	120
Other Pacific	139
Russian Federation	90
Latin America	54
Asia (excluding China)	202
China	216
Africa	175

Source: IEA (2009).

The variation in CO_2 intensity is explained by the different energy sources and energy mixes utilized in different regions and countries. For example, regions such as Asia and Africa, and countries like Australia that rely on coal as their dominant source of energy for electricity production, have on average higher CO_2 emissions compared with Latin America and New Zealand with lower CO_2 emissions per MJ produced owing to the higher proportion of electricity that is based on renewable resources like hydroelectric power which are recognized to be carbon neutral.

3. EMISSIONS RELATED TO TRANSPORT

The food sector is transport-intensive – large quantities of food are transported in large volumes and over long distances. This can sometimes be of significance but, in terms of the overall contribution to the life cycle carbon footprint of a product, most LCA studies have found that the contribution of transport is relatively small. The carbon implications of food transport is not only a question of distance; a number of other variables, such as transport mode, efficiency of transport loads and the condition of infrastructure (road quality), fuel type, etc., are important determinants of the carbon intensity of products.

The efficiency of different transport modes varies considerably. Transport modes differ significantly in energy intensity and hence GHG emissions. Air transport has a very high climate change impact per tonne transported, whereas sea transport is relatively efficient. Long-distance transport by ship is very energy efficient, with estimates between 10 and 70 g CO_2 per tonnes-km, compared with estimates of 20-120 and 80-250 g CO_2 per tonnes-km for rail and road, respectively (Marintek, 2008). Poor road infrastructure has an impact on the emission per unit product transported because it increases fuel consumption. Cederberg *et al.* (2009) found that, in Brazil, due to generally poor road conditions, the consumption of diesel was estimated to be 25 percent higher than under normal road conditions. Different loads also affect the efficiency of utilization of transport per unit of product. Larger loads transported for longer distances are more efficient than lighter loads transported over shorter distances.

During transportation, food also often requires refrigeration which increases the use of energy and also introduces leakage of refrigerants into the GHG emissions equation (refrigerants are often high in climate impact).

Emissions related to transport were estimated for the different phases, that is, transportation of live animals from the farm to the slaughter plant and transportation of the processed product from plant to retail centre for distribution. In the case of international trade, emissions were calculated for transport from slaughter plant to the port of export to the retail point for distribution. In an effort to estimate the contribution of international freight transport to GHG emissions, we combined data on trade flows, transportation mode, transport EFs and distances.

The following sections provide a detailed description of the methodology and the assumptions used in the estimation of emissions associated with the transport of live animals and meat. For the approach on milk, a detailed description is provided in FAO's report on GHG emissions in the dairy sector published in 2010.[20]

[20] FAO. 2010. http://www.fao.org/docrep/012/k7930e/k7930e00.pdf

3.1 Transport of live animals from the farm to slaughter plant

Due to the complexity of live animal movements and data limitations, several simplifications and assumptions were made:

- *Share of animals transported to slaughter plants*: Not all animals produced are slaughtered in slaughter plants/abattoirs; slaughtering may also take place on-farm or may be carried out by local butchers within the vicinity of production and thus may not involve the transportation of live animals. For industrialized countries, it was assumed that about 98 percent of the animals are slaughtered in slaughterhouses. In developing countries, the share of animals transported to slaughter plants varied between 15 and 75 percent. A lower share was assumed for developing countries based on the assumption that slaughtering infrastructure is generally lacking and that animals are often slaughtered in closer proximity to where they are raised, with slaughter being carried out by local butchers or the household itself. Other factors taken into consideration include the importance of exports within the economy, where we assumed that key exporting developing countries such as Brazil, Argentina, Paraguay, Botswana and Namibia (due to phytosanitary requirements of importing countries) would have a higher share of animals slaughtered in slaughter plants.
- *Average distance between farm and slaughter plant*: Data on distances between the farm and slaughter plants was taken from literature for industrialized regions: an average distance of 80 km for Europe and 200 km for North America. In developing countries, due to poor infrastructure, slaughter is assumed to take place near the point of sale: an average distance of 50 km was assumed.
- *Mode of transport:* We assumed that a greater proportion of live animals was transported by road.
- *Emission intensity per kg of carcass transported*: Based on secondary data, two average coefficients were utilized in this study for two groupings of countries: 0.21 and 0.38 kg CO_2-eq per tonnes CW-km for industrialized and developing countries, respectively.

Transport emissions of livestock between the farm and the slaughter plant were calculated using the equation below:

$$GHG_{transport}^{farm\text{-}slaughterplant} = D_{farm\text{-}plant} \cdot ef_{transport} \cdot sh_{live\ animal}^{farm\text{-}plant}$$

where:

$GHG_{transport}^{farm\text{-}slaughterplant}$ = GHG emission intensity, kg CO_2-eq/kg CW-km

$D_{farm\text{-}plant}$ = average distance between farm and slaughter plant, km

$ef_{transport}$ = average EF for transport, kg CO_2-eq/kg CW-km

$sh_{live\ animal}^{farm\text{-}plant}$ = share of animals transported from farm to slaughter plant, percentage

Table D2. Emission intensity for road transport

	Winther *et al*. 2009, *(100 percent)**	Winther *et al*. 2009, *(100 percent)**	Ecoinvent, *(100 percent)**	Ecoinvent, *(70 percent)**	Ecoinvent, *(90 percent)**	AEA 2008	Cederberg *et al*. 2009
	(kg CO_2-eq/tonnes CW-km)						
Articulated lorry, max load 32 tonne							0.11
Lorry, chilled, max load 20 tonne	0.085	0.102	0.18	0.16	0.14	0.08-0.25	
Lorry, frozen, max load 20 tonne	0.073	0.099	0.19	0.17	0.15	0.08-0.25	0.145

Note: Emission intensities also include emissions related to leakage of cooling agents.
* The EFs represent the percentage of the vehicle utilized and accounts for the fact that vehicles will not be fully utilized at all times.
Source: SIK (2010).

3.2 Transport and distribution of meat from processing plant to retail point

The calculation of GHG emissions associated with meat transport included the transport of meat from slaughter plant to a retail distribution point. Transport and distribution emissions sources comprise emissions from fuel combustion during transport, as well as emissions from energy consumption for refrigeration and refrigerant leakage from chilled vehicles or container ships. Two modes of transport were considered in this phase: refrigerated road transport and marine transport.

Road transport. Refrigerated road transport covered here refers to transport between the processing plant and the domestic market and, in the case of international trade, transport from plant to port and entry port to retail distribution centre in importing country. Table D2 presents emission intensities for different modes of road transport taken from peer-reviewed studies and databases such as Ecoinvent. Average emission intensities were found to vary depending on the transport load (tonnage), transport utilization and type of product transported (chilled or frozen).

In this study, the following average emission intensity values presented in Table D3 below were used. Regarding the transport of meat from processing plant directly to the domestic retail, we assumed that the product is transported as chilled carcass by a small vehicle with a maximum load of 20 tonnes within a minimum retail distance of 50 km.

Ocean transport. In 2005, about 6.5 and 0.97 million tonnes of beef and lamb were traded globally (FAOSTAT, 2012). Emissions from the international trade of meat were calculated on the basis of the amount and type of product traded, distances

Table D3. Average emissions intensities associated with road transport from plant to retail

	Chilled	Frozen
	(kg CO_2-eq/tonnes CW-km)	
Carcass	0.18	0.20
Boneless	0.117	0.130

Source: SIK (2010).

between the slaughterhouse and retail centre, and the average GHG emission per kg of product transported.

- *Trade:* A trade matrix was developed based on FAOSTAT trade flow data in order to determine key exporters (Table D4 as an example), destinations and quantities traded. This analysis covers almost 85 percent of the total amount of beef and lamb traded globally. A distinction is made between the type of meat traded (whether carcass or boneless) because it has implications for the amount of energy used for refrigeration during transportation and consequently CO_2 emissions.
- *Distance:* Distances were estimated between the major exporting and importing ports and it was assumed that the traded product was destined to major cities which are key population and consumption hubs. Emissions were calculated for the average distance for transport between the exporting country and importing country (port to port) and the transit distance inside the importing country to main retail centre. Distance matrices were estimated from http://sea-distances.com/index.htm and http://www.distances.com/.
- *Vessel size:* It was assumed that smaller ships are utilized for shorter distances (e.g. transport of products within regions) and larger ships for longer distances such as inter-continental trade. Table D5 presents emission intensities for ocean transport taken from secondary sources and demonstrates the variation in emission intensity for different vessel sizes.

Table D4. International trade in beef, 2005

Key exporters	tonnes
Brazil	1 285 805
Australia	991 945
USA	439 862
Ireland	363 372
Netherlands	351 757
New Zealand	344 289
Germany	335 044
Canada	323 729
Argentina	297 091
Uruguay	249 609
Total	**6 316 672**

Source: FAOSTAT (2011).

Table D5. Emission intensity for ocean transport

Container ship	Winther *et al.* (2009)	AEA (2008)	Cederberg *et al.* (2009)	Ecoinvent
	(kg CO_2-eq/tonnes CW-km)			
Large, chilled/frozen	0.037	0.018	0.014	0.011
Small, chilled/frozen	0.056		0.061	0.043

Note: Emission intensities also include emissions include related to leakage of cooling agents.
Source: SIK (2010).

Based on secondary data, the average emission intensities applied were 0.025 and 0.05 kg CO_2-eq per tonne product (CW) transported per km for large and small container ships transporting carcasses, respectively and 0.014 and 0.029 kg CO_2-eq per tonne CW per km for large and small container ships transporting bone-free meat.

To manage the versatile nature and complexity of trade flows, we only accounted for trade from and to the most significant trading partners.

4. EMISSIONS RELATED TO SLAUGHTER AND PRIMARY PROCESSING OF MEAT

GHG emissions assessed here include emissions from the direct inputs of energy in the slaughter and primary processing of meat and milk, as well as the GHG emissions related to use and leakage of refrigerants. The meat sector also produces a range of co-products including by-products such as bones, blood, fat, offal, feather, etc. Due to the lack of data on the total amount of raw material rendered, this analysis does not take into account emissions associated with the co-products.

Average energy use per kg of carcass weight during slaughter was based on studies from Sweden (Anon, 2002), Denmark, Finland and Spain (Lafargue, 2007) and the European Union (Ramirez *et al.*, 2006). Due to the limited data on energy use during this phase, in this study we assumed an average value of 1.4 MJ/kg CW and 4.5 MJ/kg CW for beef and lamb, respectively. Slaughterhouse emissions were calculated by combining this average value with the average regional specific CO_2 emissions per MJ of energy (taking into account regional/country electricity generating mixes) given in Table D1 to obtain the average GHG emissions per kg of carcass processed. Table D6 presents regional average emission factors for processed beef and lamb and mutton and illustrates the importance of energy source as well as the energy intensity associated with the processing of different meat. Compared with beef, processing of lamb and mutton on average has higher emission intensity per kg product processed because of the high energy intensity of the process (4.5 MJ/kg CW) and, when combined with high emitting energy sources such as coal, the emission intensity is high as is the case in Australia (Table D6).

Table D6. Regional emission factors for processing of beef and lamb

Region	Beef	Lamb and mutton
	(kg CO_2-eq/tonnes CW-km)	
EU27	0.14	0.45
North America	0.20	0.64
Australia	0.40	1.14
New Zealand	0.12	0.38
Japan	0.17	0.54
Other Pacific	0.20	0.63
Russian Federation	0.13	0.41
Latin America	0.07	0.24
Asia (excluding China)	0.30	0.91
China	0.30	0.97
Africa	0.25	0.78

Source: Authors' calculations.

5. EMISSIONS RELATED TO PRODUCTION OF PACKAGING MATERIALS

Packaging is a fundamental element of almost every food product and a vital source of environmental burden and waste. The type of packaging used also influences transport efficiency because it has its own weight but also because it affects the weight/volume ratio of the product. Two types of packaging can be distinguished: primary packaging and secondary packaging. Primary packaging is packaging closest to the product and often follows the product all the way to the consumer. Secondary packaging is used to assemble together primary packaging to shelter the product during transport and make it possible to transport more of the product at a time. The climate impact of packaging is one of the least studied aspects within the food chain. Due to the lack of data on the global variations in packaging of meat, this study applied 0.05 kg CO_2-eq per kg CW for both primary and secondary packaging from slaughter-plant to retail.

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APPENDIX E

Emissions related to energy use

This appendix presents the approach and coefficients applied in this study for estimating GHG emissions from direct on-farm energy use (non-feed related) and embedded energy in farm buildings and equipment. Direct and indirect emissions were estimated for all ruminant species; a general approach is used for all species with a few modifications taking into account differences between production typologies and species, but also between herd (dairy and beef).

1. INDIRECT (EMBEDDED ENERGY): EMISSIONS RELATED TO CAPITAL GOODS

Capital goods including machinery, tools and equipment, buildings such animal housing, forage and manure storage are a means of production. Though not often considered in LCAs, capital goods carry with them embodied emissions associated with manufacture and maintenance. These emissions are primarily caused by the energy used to extract and process typical materials that make up capital goods such as steel, concrete or wood. This assessment focuses on the quantification of embedded energy in capital goods including farm buildings (animal housing, feed and manure storage facilities) and farm equipment such as milking and cooling equipment, tractors and irrigation systems.

To determine the effective annual energy requirement, the total embodied energy of the capital energy inputs was discounted and we assumed a straight-line depreciation of 20 years for buildings, 10 years for machinery and equipment and 30 years for irrigation systems. A simplified approach was adopted in the calculations; emission coefficients were defined for the dairy cattle sector and these were then extrapolated to the beef cattle, buffalo and small ruminant sector.

1.1 Farm infrastructure

Emissions of a representative set of farm buildings were calculated from typical material of building components, including steel, concrete and wood used in the construction of animal housing, manure storage and feed storage facilities. Data related to the density of the building material was taken from various sources and literature.

- *Animal housing*: Five different levels of housing were defined with varying degrees of quality and emissions related to these were calculated (Table E1). These five housing types were then distributed across the different production systems (grassland and mixed), AEZs (arid, humid and temperate), and country grouping [OECD, least developed countries (LDCs) and other developing countries] based on the level of economic development. The percentage allocated for the different values of housing was based on two criteria: (i) livestock density in the two production systems and three AEZs based on number of adult females; and (ii) the average milk yield per cow per year. Tables E2 and E3 illustrate the allocation of embedded energy in small ruminant housing in arid zones in OECD countries and the average emission factor (accounting for depreciation).

Table E1. Typology of animal housing considered in this assessment

Level of investment and definition	Characteristics			Production system
	Floor, foundation, walls	**Roof, roof-frame**	**Supports**	
High: *high technology and use of high quality materials*	Material: concrete	Material: steel	- Stanchions - Columns - Rafters	- Industrial units - Peri-urban - Fattening units
Average: *intermediate level of technology and use of good quality materials*	Without walls; or ½ walls (concrete)		- Stanchions - Columns - Rafters	- Peri-urban - Fattening units - Mixed systems
Low	No walls, floor not paved	Material: steel	Material: Steel	Mixed systems
Very low	Cement floor or unpaved floor (dirt) No walls	Material: steel for roof	Local/hand-made material e.g. wood for columns/ rafters	- Mixed systems - Peri-urban
Nil: *situation with no housing or existing shelter such as kraals made from local materials (wood, manure) and no embedded energy involved.*				

Source: Authors.

Table E2. An example of a life cycle inventory for a high investment structure for small ruminants

Material	Structure	GWP_{100} *(kg CO_2-eq[1])*	Quantity of material per unit *(kg of material/25 kg LW - AFSR[2])*	Emission intensity *(kg CO_2-eq/25 kg LW AFSR[2])*
		A	B	C= A/B
Concrete	Floor	262.61	0.10	26.3
Concrete	Support – foundation	262.61	0.03	6.8
Steel – structural	Support – stanchions	1.79	0.52	21.0
Steel – structural	Roof frame – rafters	1.79	0.89	9.6
Steel – structural	Roof frame – purlins	1.79	1.04	3.1
Bricks – concrete	Walls	262.61	0.03	4.7
Galvanized metal – shed	Roof	1.79	1.07	4.1
Total				**83.7**

[1] Data taken from Ecoinvent database.
[2] AFSR: Adult Female Small Ruminant.
Source: Authors' calculations.

Table E3. Allocation of embedded energy in housing – an example for small ruminants in arid zones in OECD countries

Country grouping	Level of investment	Emission intensity *(kg CO_2-eq/25 kg LW AFSR[1])*	Allocation *(percentage)*	Emission factor *(kg CO_2-eq/25 kg LW AFSR[1])*
OECD	High	83.7	20	16.7
	Average	76.7	20	15.3
	Low	47.0	50	23.5
	Very low	10.5	-	0.0
	Nil	0.00	10	0.0
Total			100	**55.6**
Depreciation (20 years)				**2.8**

[1] AFSR: Adult Female Small Ruminant.

- *Manure storage:* The calculation for energy embodied in manure storage facilities was based on a similar methodology and allocation technique outlined above. As capital investment, only a platform of concrete was considered and calculated as a percentage of the floor-surface of the standard shelter (25 percent on 90 days and 50 percent on 180 days of manure storage). The period of manure storage considered includes 90 days in arid and humid areas and 180 days in temperate days in both grassland and mixed systems. Although liquid manure storage plays an important role in industrialized regions particularly for dairy, only solid manure storage was considered for this assessment.
- *Feed storage:* The calculation for energy embodied in feed storage facilities was based on a similar methodology and allocation technique outlined for housing and manure storage. The period of feed storage considered includes 90 days in arid and humid areas and 180 days in temperate days in both grassland and mixed systems. Due to their importance in a majority of countries, only hay and straw were used as the basis for feed density. The required volume of storage capacity was calculated on the basis of roughage requirements (based on 2 percent intake of DM) and the Bulk Specific Weight and Density[21] for hay and straw. The quality of the feed storage was assumed to be similar to the animal housing infrastructure.

1.2 Farm equipment

Emissions embodied in farm equipment were calculated on the basis of the five levels of farm infrastructure (ranging from nil to high), with allocation criteria similar to those outlined for farm infrastructure. For these calculations, farm equipment was divided into three categories: tractors, tractor implements and hand tools; milking and milk storage equipment; and irrigation facilities. Emissions related to steel were derived from the Ecoinvent database.

- *Tractors, implements and hand tools*: The calculation for energy used in the manufacture of tractors and tractor implements and tools is related to the number of tractors used per hectare; an average weight of steel per hectare based on Dyer and Desjardins (2005); and the stocking rate of adult females per hectare. It is assumed that in areas with over 1 000 ha per tractor, the use of hand tools is prevalent and for these situations we estimated 5kg of hand tools.
- *Milking and storage/cooling equipment*: Equipment taken into account includes bulk tanks and cans, post bars, vacuum pump, pipelines, plate cooler units. Table E4 presents the milking and storage/cooling equipment considered in this study.
- *Irrigation systems*: Two basic types of irrigation systems were considered: border strip and spray irrigation and were applicable only to the high and average level of investment farm. Due to the lack of more recent data, the calculation for energy embodied in irrigation systems is based on the approach used by Wells (1998).

[21] This is a measurement of a feed's mass (weight) per unit volume of space the feed occupies; the standard unit is kg/m^3.

Table E4. Milking, cooling and storage equipment considered in this assessment

Equipment	Description
Coolers	Medium-scale herd composed of 40 cows producing 20 l/day (milked twice a day); using a tank of 1600 litres Small-scale herd composed of 14 cows producing 20 l/day (milked twice a day); using 10 cans of 60 litres each
Post bars	Medium-scale herd – 4+4 posts steel made Medium-scale herd – 2 posts wood made
Pipeline	Medium-scale herd – double pipeline set suspended over the central corridor
Milking vacuum pump	Medium-scale herd – consider an average typology: 2 mobile and 1 fixed floor
Cooling system	Medium-scale herd – consider an average value among low, medium, high

Source: Authors.

Table E5 presents average emission factors for embedded energy for on-farm capital goods in dairy cattle production. For beef cattle and buffalo, we took a simplified approach by applying 50 percent of the EF coefficient calculated for dairy cattle. Emission factors used for small ruminant dairy are presented in Table E6 and a similar approach of applying 50 percent of the EFs to small ruminant meat herds was adopted.

2. DIRECT ENERGY: EMISSIONS RELATED TO ON-FARM ENERGY USE

On-farm energy in ruminant production relates to the use of energy for milking, milk pumping, on-farm cooling of milk, ventilation, heating and lighting, water heating, watering and feeding of animals.

Various studies have estimated the amount of direct energy used on farm (Barrington *et al*., 1999; Dalgaard *et al*., 2000; Cederberg and Mattsson, 2000; ADAS, 2000; Haas *et al*., 2001; Wells, 2001; Ludington and Johnston, 2003; Barber and Pellow, 2005; Casey and Holden, 2005; Dyer and Desjardins, 2006; Saunders and Barber, 2007; DEFRA, 2007a, 2007b; Schils *et al.*, 2007; FEC, 2008; Horndahl, 2008; DairyCo, 2009; Thomassen *et al*., 2008; CAFRE, 2009; Bestfootforward (personal communication, 2010); ATTRA, 2010; Williams *et al*., 2010; Rotz *et al.*, 2010). Based on these studies, it is estimated that the average energy use is 0.219 kWh/kg raw milk.

It is however difficult to make an accurate estimate of the average energy use for these individual farm processes as well as the type of energy used because of the lack of disaggregated data. However, four studies (Bestfootforward (personal communication, 2010); Ludington and Johnston, 2003; DEFRA, 2007b); and Thomassen *et al*., 2009) provide a breakdown by source, which indicates that 38 percent of total direct energy consumed on-farm is electricity and 62 percent non-electricity. Using the results above (i.e. total direct energy use is 0.219 kWh/kg raw milk, which is split 38:62 electricity: non-electricity) and assuming that the main non-electricity use is diesel, the emissions can be calculated, see Table E7. The two coefficients (0.083 and 0.135 kWh/kg milk) are used in the calculation of the EFs for on-farm direct energy use.

Countries were ranked by milk yield, then categorized into five groups (representing the five categories of dairy farm mechanisation: High, Average, Low, Very low, Nil). Energy use will vary between these five levels. It was assumed that the

Table E5. Average emission factors for embedded energy for dairy cattle

Grouping	System	Capital goods	Arid (kg CO_2-eq/ 100 kg LW)	Humid (kg CO_2-eq/ 100 kg LW)	Temperate (kg CO_2-eq/ 100 kg LW)
OECD	Grassland based	Buildings	4.42	4.72	9.03
		Machinery & Implements	13.78	16.22	28.16
	Mixed based[1]	Buildings	4.89	5.08	9.55
		Machinery & Implements	16.22	19.03	30.23
LDC countries	Grassland based	Buildings	0.68	0.68	0.83
		Machinery & Implements	1.35	1.35	1.35
	Mixed based[1]	Buildings	1.33	1.33	1.89
		Machinery & Implements	1.98	1.98	1.98
Non-OECD	Grassland based	Buildings	1.71	2.31	3.32
		Machinery & Implements	2.94	5.85	4.04
	Mixed based[1]	Buildings	2.38	3.04	6.64
		Machinery & Implements	3.56	6.62	18.55

[1] Includes landless systems
Source: Authors' calculations.

Table E6. Average emission factors for embedded energy for dairy sheep and goats

Grouping	Production system	kg CO_2-eq/25 kg LW	kg CO_2-eq/100 kg LW
LDC	Arid	1.00	0.04
	Humid	0.82	0.03
	Temperate	0.73	0.03
OECD	Arid	5.65	0.23
	Humid	5.05	0.20
	Temperate	6.76	0.27
Other developing	Arid	2.01	0.08
	Humid	2.62	0.11
	Temperate	6.01	0.24

Source: Authors' calculations.

Table E7. Total on-farm direct energy use and associated emissions for high level dairy farms

Category	Rate of energy use (kWh/kg milk)	Emissions (kg CO_2-eq/kWh)	Emissions (kg CO_2-eq/kg milk)
Electricity	0.08	0.54	0.05
Non-electricity	0.14	0.27	0.04
Total			**0.08**

Source: Authors' calculations.

Table E8. Emissions from direct energy for different levels of mechanization

Category	Rate of electricity use	Rate of non-electricity	Emissions from electricity	Emissions from non-electricity	Total emissions from direct energy
	(kWh/kg milk)		*(kg CO_2-eq/kg milk)*		
High	0.08	0.14	0.05	0.04	0.08
Average	0.08	0.07	0.05	0.02	0.06
Low	0.08	0.03	0.05	0.01	0.05
Very low	0.00	0.01	0.00	0.00	0.00
Nil	0.00	0.00	0.00	0.00	0.00

Source: Authors' calculations.

Table E9. Emission factors for direct on-farm energy use for dairy cow milk production in OECD and non-OECD countries

Region	Electricity EF[1]	Default global EF	Grassland			Mixed		
	($kgCO_2/kWh$)							
Unadjusted EF			**Arid**	**Humid**	**Temperate**	**Arid**	**Humid**	**Temperate**
Europe (Unadjusted EF)			0.071	0.072	0.074	0.072	0.074	0.074
Developing countries			0.020	0.020	0.020	0.027	0.027	0.027
Non-OECD			0.038	0.044	0.05	0.045	0.054	0.062
Adjusted emission factors								
EU-27	0.36	0.54	0.059	0.060	0.061	0.060	0.061	0.061
OECD-Europe	0.34	0.54	0.058	0.059	0.060	0.059	0.060	0.060
USA	0.54	0.54	0.071	0.072	0.074	0.072	0.074	0.074
Canada	0.19	0.54	0.048	0.049	0.050	0.049	0.050	0.050
OECD North America	0.50	0.54	0.068	0.069	0.071	0.069	0.071	0.071
Australia	0.90	0.54	0.094	0.095	0.098	0.095	0.098	0.098
Japan	0.44	0.54	0.064	0.065	0.067	0.065	0.067	0.067
South Korea	0.46	0.54	0.066	0.067	0.068	0.067	0.068	0.068
New Zealand	0.21	0.54	0.049	0.050	0.051	0.050	0.051	0.051

[1] IEA (2010).

Source: Authors' calculations.

rate of electricity use would be the same for high, average and low systems, where milking activities are largely mechanized. It was further assumed that no electricity is used in very low and nil level systems. For the allocation across the five categories, the median adult female (ADF) weight and milk yield were used and the milk yield per kg of ADF calculated for each category. Table E8 presents the emission intensity of milk from direct on-farm energy use for the different levels of mechanization.

On-farm energy use was then adjusted to reflect the variations across farming systems in terms of level of mechanisation and energy use efficiency. It was assumed that 50 percent of OECD emissions are from electricity. The EFs for OECD countries were adjusted to take into account variations in the amount of CO_2 emitted per kWh electricity. It was assumed that non-OECD countries do not use mains electricity, and standard emission factors are used for diesel/petrol (which means

Table E10. Regional emission factors for direct on-farm energy use

Region	Beef: grassland based	Beef: mixed	Sheep and goats: meat
	kg CO_2-eq/kg LW		
Europe			
EU27	0.18	0.21	0.33
OECD-Europe	0.17	0.21	0.33
Non-OECD Europe	0.07	0.09	0.19
North America			
USA	0.24	0.29	0.34
Canada	0.12	0.15	0.31
Other	0.22	0.27	0.34
Pacific			
Australia	0.36	0.42	0.38
Japan	0.20	0.24	0.33
South Korea	0.21	0.25	0.34
New Zealand	0.12	0.16	0.31
OECD Pacific	0.22	0.27	0.34
Pacific average	0.00	0.00	0.34
Non-OECD Pacific	0.00	0.00	0.17
Former Soviet Union	0.07	0.09	0.16
Latin America			
Brazil	0.07	0.09	0.15
Other	0.07	0.09	0.16
Asia			
India	0.07	0.09	0.19
China	0.07	0.09	0.18
Thailand	0.07	0.09	0.17
Other	0.07	0.09	0.18
Africa	0.07	0.09	0.18
Middle East	0.07	0.09	0.18

Source: Authors' calculations.

that emissions in some countries, e.g. Brazil, will be overestimated). Table E9 presents the adjusted EFs for direct on-farm energy use for milk production in OECD countries and EFs for non-OECD countries.

Direct on-farm energy for non-dairy herd (beef cattle, buffalo, small ruminant meat herd/flock): Direct energy use is associated primarily with the handling of manure, bedding and feed, which are dependent on the system (i.e. grass or mixed) and the level of mechanization, rather than the climate. Therefore it was assumed that the energy use for a given mechanization level and system is independent of climate. We assumed that minimal direct energy use is associated with meat production was assumed –in developing regions - low levels housing and mechanized feed, bedding manure handling. Table E10 presents regional EFs for direct on-farm energy use for ruminant meat production used in this assessment.

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APPENDIX F

Relative value of slaughter by-products and effect on allocation of emissions

The main edible product from slaughtered animals is meat, but slaughterhouses also produce a whole range of by-products (organs, hide, blood, etc.). Between 30 and 60 percent of animal weight, depending on the species, does not end up as meat for human consumption.

Little documented information is usually available on the marketing of by-products from abattoirs but it is generally considered that they constitute a crucial part of profitability, with a more than significant share of the margin. They are subdivided into edible and non-edible materials.

1. EDIBLE BY-PRODUCTS

The main edible by-products of a slaughtered animal are offal, also known as variety meat or organ meat. Offal is divided into red (heart, livers, kidneys, lungs, tongue, cheek meat and deboned head trimmings) and white (intestine, stomachs, sweetbread [thymus and pancreas] and brain). Edible by-products can also be blood and fats that are fit for human consumption, and used in further processed products such as sausages.

According to a survey in the French meat industry, all edible materials from the carcass, including meat and offal, account for 45 percent of the live weight of an adult cattle (see Table F1). By-products are therefore 55 percent of the animal live weight. This is consistent with the results of a study on yields of by-products in various breeds of cattle slaughtered in Texas in 1989 (Terry *et al.*, 1990). It is also

Table F1. Beef cattle products and by-products, by type of use

Use of animal products and by-products (ABP)	Beef *(percentage of LW)*
EDIBLE	
Meat, offal, blood and fats for human consumption	45
INEDIBLE	
ABP withdrawn for sanitary reasons, SRM*, wastes	10
Protein, blood and fats (pet food, animal feed, drug industry)	20
Bones (feed industry, glue, gelatine)	8
Skins & hides (leather)	6
Digestive tract content (compost/fertilizers)	10
Lost due to carcass chilling and drying	1

* Specified risk material with regard to BSE (brain, eye, medulla, etc.).
Source: FranceAgriMer. 2012. *Observatoire des coproduits* (based on a survey of 40 meat plants in France).

consistent with the results of a more recent USDA study (Marti *et al.*, 2011), that estimated total by-products at 44 percent of total live weight, but did not include the digestive tract content (approximately 10 percent).

Edible by-products, mainly offal, account for around 12 percent of adult cattle live weight (Ockerman and Hansen, 2000). Human consumption of offal varies by culture and region, but can be found almost everywhere, in developed as well as in developing countries. Following animal health crises, such as the outbreak of *bovine spongiform encephalopathy* (BSE) in 1996, and the ban on the use of these products by a number of countries, the global offal market accounts for 15 to 20 percent of production.

World trade of bovine offal is estimated at one million tonnes per year. Asia (in particular China and Japan) is the main outlet for bovine offal, and is far from being self-sufficient, with 40 percent of global imports. Russian Federation doubled its imports in the past ten years, importing more than 100 000 tonnes of beef offal today, despite the ban on U.S. beef in 2004 due to BSE. Other significant importers include Egypt and Central Africa.

Offal exporters are the main beef exporters: United States (27 percent), Australia (14 percent), Argentina (12 percent) and Brazil (9 percent). In other countries, offal is often sold locally.

2. NON-EDIBLE BY-PRODUCTS

For adult cattle, non-edible by-products represent on average 55 percent of the total live weight. Material to be eliminated, such as by-products withdrawn for sanitary reasons (e.g. liver with flukes), specified risks materials and wastes from the first water treatment, account for 10 percent of the animal weight and are a cost for the meat plant.

Hides and skin constitute the most profitable non-edible by-products of the meat industry, with about 6 percent of the animal weight and sometimes up to 75 percent of the by-products value (Marti *et al.*, 2011). They are also the most internationally traded by-products, with Italy and Turkey as major outlets for many exporters.

In terms of weight, the most important group of non-edible by-products (accounting for 20 percent of total live weight) is constituted by floor trimmings, blood and fats used mostly for pet food but also for animal feed (processed animal protein, like meat and bone meal), or in the drug or cosmetic industry. Bones (8 percent of the weight) often go through rendering with this category to produce processed animal protein. They can also be used to produce glue or gelatine that go back into the human consumption chain.

Digestive tract content is usually about 10 percent of the animal weight and is used as fertilizer or as biogas material on the meat plant to produce energy.

3. VALUE OF BY-PRODUCTS

Edible and non-edible by-products accounted for 11 percent of the total value of the carcass sold by slaughterhouses in 2011, according to a survey of 10 cattle slaughterhouses in France (Observatoire des prix et des marges, 2012). The share of by-products in the total value tends to increase over time (it was only 6 percent in 2005).

This result is consistent with a study by Terry *et al.* (1990) that estimates the value of edible and inedible by-products from cattle at 9 to 12 percent of the total live

Table F2. Total revenue from one adult cattle sold by the slaughterhouse and share of by-products

Year	Total revenue (€/kg)	All by-products value (€/kg)	All by-products (percentage)	Edible by-products value (€/kg)	Edible by-products (percentage)	Non-edible by-products (percentage)
2005	4.19	0.28	6	0.14	3	3
2006	4.39	0.29	6	0.15	3	3
2007	4.39	0.38	9	0.15	4	5
2008	4.5	0.26	6	0.16	4	2
2009	NA	NA	NA	0.17	NA	NA
2010	4.69	0.4	9	0.16	4	5
2011	4.96	0.53	11	0.16	3	8

NA: Not Applicable.
Source: Observatoire des prix et des marges, 2012; Service de Nouvelles des Marchés.

Table F3. Emissions intensity of beef with and without allocation to slaughter by-products in Western Europe

	kg CO_2-eq/kg LW
No allocation to by-products	18.8
With allocation to by-products	17.7

Source: Authors' calculations.

value. It is also consistent with Marti *et al.* (2011), who estimated that by-products added value to one steer at 10 percent in average over the period 2000 to 2011.

Nevertheless, because of consumption habits, value for edible by-products can be very different from one country to another. For example, offal like hearts or stomach has greater value on the Chinese market than on any other market.

According to data from the Rungis Wholesale Market in France, total value of offal for one adult cattle was € 52.2 in 2011, that is to say 0.16 cents per kg of total products for one animal. Offal market prices were lower in 2005, and the market is very sensitive to sanitary crises, but the contribution of edible by-products in the total revenue from slaughtered adult cattle is generally stable at about 3 to 4 percent. We estimate that the value of other edible by-products is not significant compared with edible offal.

Non-edible by-products therefore account for the rest of the revenue from by-products; about 8 percent of the total revenue in 2011, a share that has increased by 5 percent since 2005.

The global value of non-edible by-products is quite volatile and this is mainly driven by the value of hides and skins. The world skin markets drop of 2008 and 2009 is reflected in Table F2 with a decrease of non-edible by-products in 2008.

This case study in France is one of the few examples available. Results appear to be consistent with a similar study by USDA but they cannot be seen as representative on a global scale since the categories and actual uses of slaughter by-products varies greatly from region to region and in time. Furthermore, alternative types of allocation (e.g. dry mass) could be used however these require further developments.

Nevertheless, because of the similarities among Western European breeds and among European markets of animal products, we can extrapolate the results of the French case study to Western Europe. Table F3 presents the effect of allocation emissions to by-products using 6 percent as the allocation value of emissions to slaughter by-products in Western Europe.

Due to the lack of a comprehensive global data on by-products in the meat sector, the allocation of emissions to slaughter by-products could not be performed in this assessment. This may be improved in a future assessment depending on information shared by the industry and the development of harmonized methods to allocate by-products at slaughterhouses.

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