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 land-use change. 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 are also provided.

Land use, land-use change 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 soil organic carbon (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).[13] 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 (Kwon and Post, 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 and Goodale (2004) estimate that the average annual emissions from LUC is 2.2 petagram C/year, 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 that 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 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).

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, 2004a), 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

---

[13] 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).

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 g C/m^2 per year.

There is also considerable potential to sequester carbon in croplands through a range of available options 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 promote the conservation of SOM and increase C accumulation. For example, Dieckow *et al.* (2010) assessed the 17-year contribution of no-tillage crop rotations to C accumulation in the subtropical Ferralsol of Brazil and concluded that crop-forage systems and crop-based systems with legume represent viable strategies to increase soil organic C stocks. They found that alfalfa systems with maize every three years showed the highest C accumulation (0.44 tonnes C/ha/yr). The biannual 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 as well as other environmental impacts (such as non-CO_2 GHG emissions).

2. QUANTIFICATION OF CARBON EMISSIONS AND SEQUESTRATION

2.1 Changes of carbon stocks related to land-use changes

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 tracking requires a time series of data, or data collected from 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 between the simple and the more sophisticated approaches involves big tradeoffs 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 EFs to more sophisticated approaches that use detailed location-specific data and process models that fully characterize the fluxes between carbon pools.

2.2.1 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 EFs 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 provide 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 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). In order to account for these changes, areas of converted land must 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, the LUC emissions for the major carbon pools, i.e. 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 EFs, 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 for land use estimates.

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, 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 are 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 have 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

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).

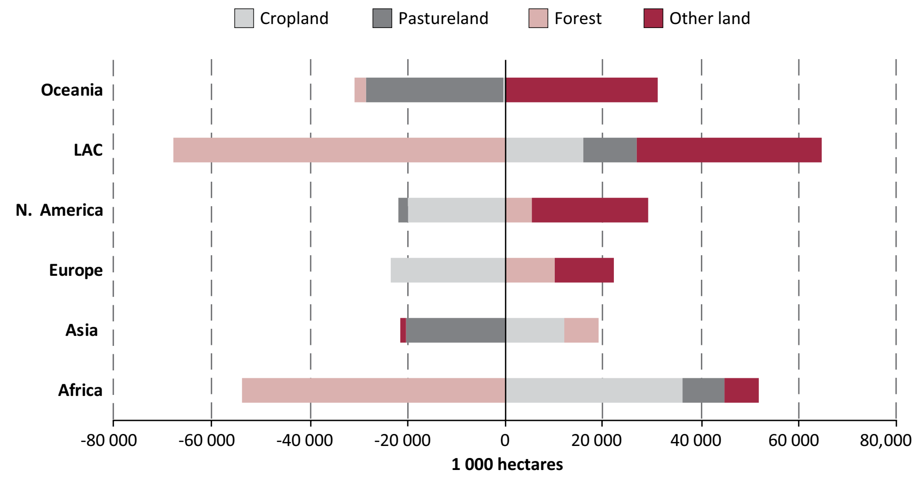
This study uses IPCC guidelines as a basis for the quantification of LUC emissions. This choice is largely based on the fact that the IPCC approach meets the UNFCCC needs for calculating and reporting of GHG emissions from LUC. The cropland part of this assessment also relies on other guidelines such 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[14]), 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

---

[14] 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.

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



Source: FAOSTAT (2012).

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

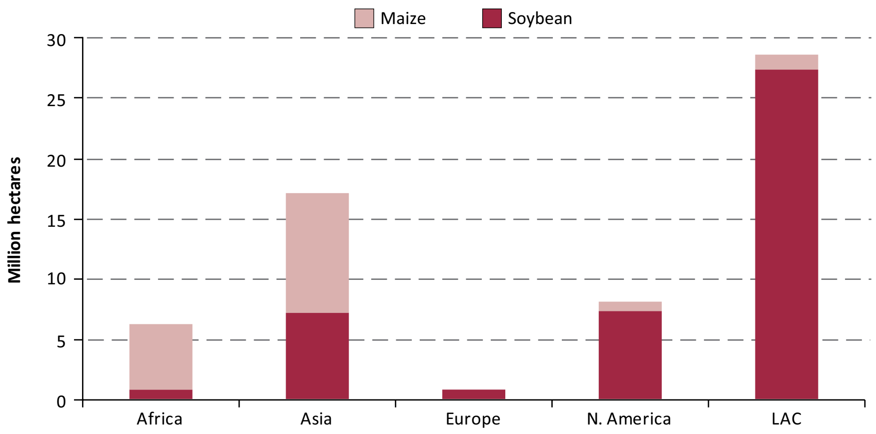
Crop	Area expansion *(1 000 ha)*	Share of global gross crop expansion *(percentage)*
Soybeans	38 110	22.6
Maize	15 620	9.2
Rapeseed	9 815	5.8
Rice, paddy	8 650	5.1
Sunflower seed	7 237	4.3
Oil palm fruit	7 205	4.3

Source: FAOSTAT (2012).

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 Latin America (reduced deforestation rates) and Asia (increasing rates), 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, latterly, biofuel policies. The expansion of soybean production is argued to be one of the major drivers of LUC, particularly 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, some other crops, such as wheat, barley, and oats, have strongly declined, which explains the apparent discrepancies with the net land conversion trends in Figure C1.

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



Source: FAOSTAT (2012).

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.

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), but 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, respectively, soybean and maize expansion in the country.

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

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).

Based on these observations the scope of our assessment was reduced 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 the key producing regions 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 possibly left as fallow land. The latter, classified as “other land” in FAOSTAT, is occupied by weeds, grasses, shrubs and, partly, by secondary forest. Under this assumption, every year roughly 2.9 million ha are converted to arable land and grassland during the period covered in this assessment. 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 also based on IPCC (2006). These 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.

Table C3. Regional sources of soybean and soybean cakes in 2005 *(percentage)*

	Brazil		Argentina		Other	
	Soybean	Soybean Cake	Soybean	Soybean Cake	Soybean	Soybean Cake
LAC	42	49	41	15	17	36
E & SE Asia	17	7	14	10	68	83
E. Europe	0	9	0	27	100	63
N. America	0	0	0	0	100	100
Oceania	0	60	0	0	100	40
Russian Fed.	5	5	0	37	95	57
South Asia	6	2	1	0	93	98
SSA	0	0	1	60	99	39
NENA	12	7	19	23	69	69
W. Europe	61	34	0	38	38	28

Source: FAOSTAT (2013).

Table C4. Main exporters of soybean and soybean cakes in 2005

	Soybean		Soybean cake	
	Exports *(Million tonnes)*	Share of global exports *(percentage)*	Exports *(Million tonnes)*	Share of global exports *(percentage)*
Argentina	20.8	37	10.0	15
Brazil	14.4	26	22.4	34
United States of America	5.1	9	25.7	39
India	4.8	8	0.0	0
Paraguay	0.8	1	3.0	5

Source: FAOSTAT (2013).

Table C5. Land-use change emissions associated with soybean production

Region	Pigs	Chicken	Cattle
		(Million tonnes CO_2-eq)	
LAC	19.3	47.9	5.2
East Asia	25.3	25.1	0.9
E. Europe	2.1	0.4	0.6
N. America	0.0	0.1	0.5
Oceania	1.5	1.6	2.4
Russian Fed.	0.1	0.1	0.1
South Asia	0.0	4.5	0.0
SSA	0.0	0.5	0.0
NENA	0.0	5.6	0.2
W. Europe	36.7	23.9	19.6
World	85.0	109.6	29.6

Source: GLEAM.

In quantifying total emissions associated with the transformation of forest for soybean cultivation, LUC emissions are attributed to only those countries supplied by Brazil and Argentina with soybean and soybean cake. Table C3 gives the share of soybean and soybean cake sourced from Brazil and Argentina for each region, and Table C4 gives the main exporters.

3.2 Results for land-use change

This analysis shows that about 224 million tonnes CO_2-eq arise *per annum* from the expansion of soybean production in Brazil and Argentina to meet global demand for pigs, chickens and cattle feed. The bulk of these emissions arise in response to soybean consumption in Europe, East Asia and LAC (Table C5) which source large quantities of their soybean feed from Argentina and Brazil. The emissions estimated for the livestock sector in Western Europe are particularly high, which not only indicates a high reliance on imported soybean and soybean cake for feed, but also use of soybean with a high emission intensity, particularly because a large share is sourced from Brazil (see Table C3).

Meeting demand for pig and chicken feed accounts for 195 million tonnes CO_2-eq *per annum*, 87 *percent* of the total. This result is not surprising because of the high share of soybean in the diets of these species. Regarding the cattle sector, LUC emissions from soybean are important in Europe where it is utilized in dairy production. The results suggest that emissions are largely influenced by: (i) the quantity of soybeans and soybean cake imported from the two countries; and (ii) the share of soybean in the ration.

The results presented here are part of the ongoing process of improving the estimation of LUC emissions. In order to progress towards better methodologies, certain gaps in data and in scientific understanding need to be addressed. The following section outlines some of the challenges and investigates the influence that methodological choice can have on LUC emissions.

3.3 Sensitivity analysis and the influence of land-use change 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 C6.

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;

Table C6. Alternative approaches for soybean LUC emissions calculations

Method	Spatial allocation	Temporal allocation of LUC emissions (amortization)	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.

Brandao *et al.*, 2005). Due to the lack of knowledge of the origin of the converted 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

Table C7. Summary of land-use change emission intensity in current study: alternative approaches for soybean cake

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.

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.

3.2 Results

Effect of LUC approach on soybean LUC emission factor. Table C7 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.

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 C8). 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

Table C8. Proportion of expanded soybean area derived from each land use

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).

appear to be close over the two periods 1990-2006 and 2002-2007 (1.76 and 1.98 million ha respectively, Figure C3), 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.

Effect of LUC approach on meat and egg emission intensity. In order to test the sensitivity of the results to different soybean LUC methods, the analysis of pigs and chickens in the UK and Viet Nam was rerun with the emission intensities calculated using the different LUC approaches. Results are given in Table C9.

Within country effect. The effect of changing the soybean LUC approach on the emissions intensity of meat or eggs can vary between different combinations of system and species within the same country as a result of differences in the percentage of soybean in ration. For example, changing the soybean LUC approach results in a greater change in the emission intensity of UK broilers than layers (Table C10) as they have a greater percentage of soybean (and therefore soybean from Brazil) in their ration compared to UK layers (Table C12).

Furthermore, the relative importance of feed emissions to total emissions intensity influences the proportionate increase in emissions intensity. For example, feed emissions make up a greater proportion of the broilers total emission intensity (as they have lower manure emissions and higher feed conversion ratios), so a 10 percent change in the feed emission intensity will lead to a greater increase in broilers than layers.

Finally, soybean used in backyard systems is assumed to not be associated with LUC, so the emissions intensity within these systems therefor does not vary in response changing soybean LUC methods (Table C10 and C11).

Between country effect. Differences in the total amount of soybean that is imported affect the emissions intensity of meat/eggs when using the One-Soy approach because with this method the EF for soybean varies depending on the percentage of the soybean that is imported, rather than the specific country that it is imported from.

Figure C3.
Annual forest loss in Brazil

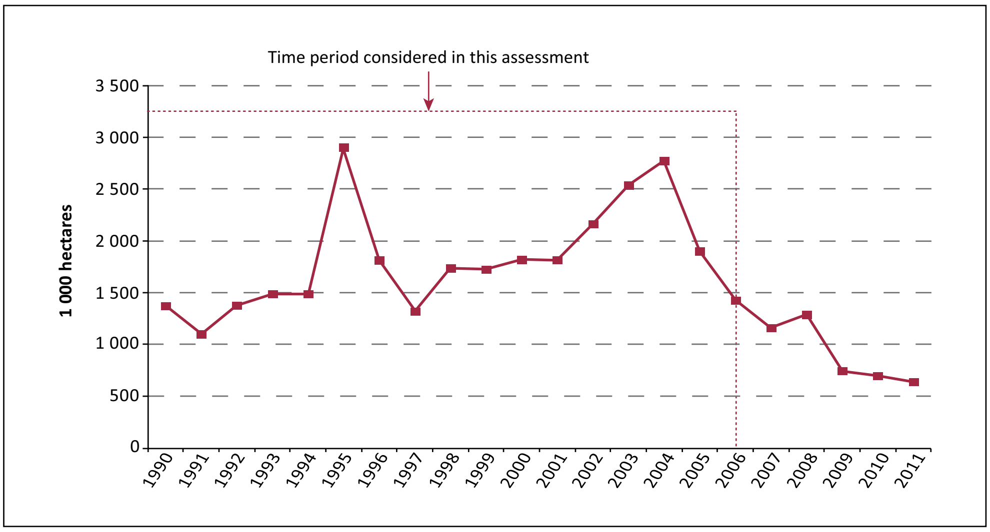


Table C9. Soybean land-use change emissions factors for the United Kingdom and Viet Nam in 2005 *(kg CO_2-eq/kg soybean DM)*

	United Kingdom		Viet Nam	
	Soybean	Soybean cake	Soybean	Soybean cake
GLEAM approach (current study)	3.6	3.2	0.5	0.5
PAS 2050-1:2012	1.8	1.6	1.5	1.3
One-Soy	3.3	3.0	2.8	2.6
Reduced time-frame	1.7	1.6	0.2	0.2

Source: Authors' calculations.

Table C10. Total emissions intensity for chickens *(kg CO_2-eq/kg meat/egg protein)*

	Broilers: UK	Layers: UK	Backyard: Viet Nam
GLEAM approach (current study)	43.5	28.5	40.3
PAS 2050-1:2012	32.9	23.5	40.3
One-Soy	41.8	27.6	40.3
Reduced time-frame	32.5	23.3	40.3

Source: Authors' calculations.

Table C11. Total emissions intensity for pigs *(kg CO_2-eq/kg meat/egg protein)*

	Industrial: UK	Intermediate: Viet Nam	Backyard: Viet Nam
GLEAM approach (current study)	68.8	48.1	57.9
PAS 2050-1:2012	57.3	52.0	57.9
One-Soy	66.9	57.6	57.9
Reduced time-frame	57.0	46.8	57.9

Source: Authors' calculations.

Table C12. Proportions in the ration of soybean and soybean imported from Brazil and Argentina *(percentage)*

	Soybean in ration	Soybean imported from		Soybean in ration from	
		Brazil	Argentina	Brazil	Argentina
Industrial pigs: UK	22	47	7	10	1
Intermediate pigs: Viet Nam	11	3	29	0	3
Backyard pigs: Viet Nam	1	0	0	0	0
Broilers: UK	28	47	7	13	2
Layers: UK	15	47	7	7	1
Backyard chickens: Viet Nam	1	0	0	0	0

Source: GLEAM, FAOSTAT (2012) and authors' calculations.

Differences in where the soybean is imported from, i.e. the amounts imported specifically from Brazil and Argentina, affect the emissions intensity of meat/eggs when using the GLEAM and PAS 2050 methods because with these methods the soybean EF varies depends on where the soybean is produced. Using GLEAM, Brazilian soybean has a higher EF than Argentinian (Table C7), so systems that have significant amounts of Brazilian soybean in their ration (such as UK broilers and industrial pigs – see Table C12) will have higher LUC emissions (Table C10 and C11). However, using the PAS 2050 method Argentinian soybean has a higher EF than Brazilian, leading to a higher emission intensity for intermediate pig meat in Viet Nam than under the GLEAM method (see Table C11).

In addition to the method used to calculate the soybean EF, the LUC emissions per kg of meat/eggs also depends on national differences in the amount of soybean in the ration and feed conversion ratios. Therefore, although two countries may import the same total amount of soybean, and the same amounts from Brazil and Argentina, the same species and system may still have quite different soybean LUC emissions.

4. COMPARISON WITH OTHER STUDIES

The emission intensity for LUC per kg of soybean and soybean cake calculated in this study are compared to other studies in Table C13. The emission intensity used in this study is higher than some other studies, but within the overall range.

The emission intensity of soybean is highly dependent on the method and assumptions used to calculate it (Flysjo *et al.,* 2012). Variation arises from differences in:

- The calculation of C losses in soil and vegetation (above- and below-ground);
- The quantification of land-use transitions, i.e. how much of the LUC can be attributed to cropping;
- The ways in which the LUC emissions are allocated to specific crops. Emissions can be allocated in different ways, such as: (a) the crops grown in the country/region where the LUC has occurred; (b) all expanding crops grown in the country/region where LUC has occurred; (c) all crops grown globally. These different allocation methods can lead to variations in the emissions per kg of crop;
- The time period over which emission are allocated.

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.

Table C13. Soybean land use-change emissions per unit of output and hectare

Study	Area covered by study	Emissions	*Converted/all soybean/all crops
FAO (2010a)	Argentina	1.04 kg CO_2-eq/kg soybean	all soybean
FAO (2010a)	Brazil	7.69 kg CO_2-eq/kg soybean cake	all soybean
FAO (2010a)	Brazil	8.54 kg CO_2-eq/kg soybean cake	all soybean
FAO (2010a)	Brazil	12.81 kg CO_2-eq/kg soybean cake	converted
FAO (2010a)	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 Flysjö *et al.* (2012)
Leip *et al.* (2010) mix>soybean	South America	3.10 kg CO_2-eq/kg soybean cake	all soybean Cited in Flysjö *et al.* (2012)
Leip *et al.* (2010) forest>soybean	South America	10.00 kg CO_2-eq/kg soybean cake	all soybean Cited in Flysjö *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 and Freire (2011)	Low (Argentina)	~0.5 kg CO_2-eq/kg soybean	converted
Castanheira and 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 (2010a)	Brazil – deforestation	37.00 kg CO_2-eq/ha	converted
FAO (2010a)	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.

REFERENCES

Armentano, T.V. & Menges, E.S. 1986. Patterns of change in the carbon balance of organic soil-wetlands of the temperate zone. *Journal of Ecology,* 74: 755–774.

Audsley, E., Brander, M., Chatterton, J., Murphy-Bokern, D., Webster, C. & Williams, A. 2009. *How low can we go? An assessment of greenhouse gas emissions from the UK food system and the scope to reduce them by 2050.* Food Climate Research Network and WWF, UK.

Audsley, E., Angus, A., Chatterton, J., Graves, A., Morris, J., Murphy-Bokern, D., Pearn, K., Sandars, D. & Williams, A. 2010. *Food, land and greenhouse gases. The effect of changes in UK food consumption on land requirements and greenhouse gas emissions.* The Committee on Climate Change, London.

Bickel, U. & Dros, J. M. 2003. *The impacts of soybean cultivation on Brazilian ecosystems.* Bonn, AIDEnvironment-WWF): p 33. (available at http://assets.panda.org/downloads/impactsofsoybean.pdf).

BSI. 2008. *PAS 2050:2008. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services.* British Standards Institution, UK.

BSI. 2012. *PAS 2050-1:2012. Assessment of life cycle greenhouse gas emissions from horticultural products Supplementary requirements for the cradle to gate stages of GHG assessments of horticultural products undertaken in accordance with PAS 2050.* British Standards Institution, London.

Brandao, A.S.P., Castro de Rezende, G. & Da Costa Marques, R.W. 2005. *Agricultural growth in the period 1999–2004, outburst in soybeans area and environmental impacts in Brazil.* Discussion Paper No. 1062. Instituto de Pesquisa Econômica Aplicada, Brasilia.

Carvalho, G.O., Nepstad, D., McGrath, D., Vera Diaz. M. del C. & Barros, A.C. 2002. Frontier expansion in the Amazon. Balancing development and sustainability. *Environment Sci. Policy Sustainable Dev.*, 44: 32–42.

Castanheira, E. & Freire, F. 2011. *Life-cycle greenhouse gas assessment of soybeans.* Paper presented at Life-Cycle Management Conference, 2011, Berlin.

Cederberg, C., Persson, M.U., Neovius, K., Molander, S. & Clift, R. 2011. Including carbon emissions from deforestation in the carbon footprint of Brazilian beef. *Environmental Science & Technology*, 45 (5): 1773–1779.

Coleman, K., Jenkinson, D.S., Crocker, G.J., Grace, P.R., Klir, J., Korschens, M., Poulton, P.R. & Richter, D.D. 1997. Simulating trends in soil organic carbon in long-term experiments using RothC-26.3. *Geoderma* 81: 29–44.

Conant, R.T. & Paustian, K. 2002. Potential soil carbon sequestration in overgrazed grassland ecosystems. *Global Biogeochemical Cycles,* 16(4): 1143.

Costa de Campos, B., Carneiro Amado, T. J., Bayer, C., Nicoloso, Rodrigo da Silveira, & Fiorin, J.E. 2011. Carbon stock and its compartments in a subtropical oxisol under long-term tillage and crop rotation systems. *Revista Brasileira de Ciência do solo*, 35(3):805–817

Del Grosso, S.J., Parton, W.J., Mosier, A.R., Hartman, M.D., Brenner, L., Ojima, D.S. & Schimel, D.S. 2001. Simulated interaction of carbon dynamics and nitrogen trace gas fluxes using the DAYCENT model. In Shaffer, J., Liwang Ma & S. Hansen, Schaffer, M. *et al.,* eds. *Modelling carbon and nitrogen dynamics for soil management,* pp. 303-332. Boca Raton, CRC Press, Florida.

Dieckow, J., Zendonadi dos Santos, N., Bayer, C., Molin, R., Favaretto, N. & Pauletti, V. 2010. No-tillage crop rotations, C sequestration and aspects of C saturation in a subtropical Ferralsol. 2010 19th World Congress of Soil Science, Soil Solutions for a Changing World. Brisbane, Australia.

FAO. 1996. *World livestock production systems: current status, issues and trends*, by Seré, C. & Steinfeld, H. FAO Animal Production and Health Paper 127. FAO, Rome.

FAO. 2010. *Greenhouse gas emissions from the dairy sector – A life cycle assessment, by* Gerber, P., Opio, C., Vellinga, T., Henderson, B. & Steinfeld, H. FAO, Rome.

FAOSTAT. 2012. FAO Statistical Database. Accessed 2012.

Falloon, P. & Smith, P. 2002. Simulating SOC changes in long-term experiments with RothC and CENTURY: model evaluation for a regional scale experiment. *Soil Use and Management*, 18: 101-111.

Fearnside, P.M. 2005. Deforestation in Brazilian Amazonia: history, rates and consequences. *Conservation Biology,* 19(3): 680–688.

Flysjö, A., Cederberg, C., Henriksson, M. & Ledgard, S. 2012. The interaction between milk and beef production and emissions from LUC - critical considerations in life cycle assessment and carbon footprint studies of milk. *Journal of Cleaner Production,* 28: 134–142.

Hart, P.B.S. 1984. *Effects of soil type and past cropping on the nitrogen supplying ability of arable soils*. Reading, UK, University of Reading.

Houghton, R.A. & Goodale, C.L. 2004. Effects of LUC on the carbon balance of terrestrial ecosystems. *In:* DeFries, R.S., Asner, G.P. & R.A. Houghton, eds. *Ecosystems and Land-use Change*. American Geophysical Union. Washington, D.C.

IPCC. 2000. *Land use, land-use change, and forestry*. Cambridge University Press, UK.

IPCC. 2006. *2006 IPCC Guidelines for National Greenhouse Gas Inventories,* prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). IGES, Japan.

Jenkinson, D.S., Hart, P.B.S., Rayner, J.H. & Parry, L.C. 1987. Modelling the turnover of organic matter in long-term experiments at Rothamsted. *INTECOL Bulletin* 15: 1-8.

Jones, M.B. & Donnelly, A. 2004. Carbon sequestration in temperate grassland ecosystems and the influence of management, climate and elevated CO_2. *New Phytologist,* 164: 423–439.

Kirschbaum, M.U.F. & Paul, K.I. 2002. Modelling C and N dynamics in forest soils with a modified version of the CENTURY model. *Soil Biology and Biochemistry*, 34: 341–354.

Lal, R. 1999. Soil management and restoration for C sequestration to mitigate the accelerated greenhouse effect. *Progress in Environmental Science*, 1: 307–326.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science*, 304: 1623–1627.

Leip, A., Weiss, F., Wassenaar, T., Perez, I., Fellmann, T., Loudjani, P., Tubiello, F., Grandgirard, D., Monni, S. & Biala, K. 2010. *Evaluation of the livestock sector's contribution to the EU greenhouse gas emissions (GGELS) - Final Report*, European Commission, Joint Research Centre.

Li, C., Frolking, S. & Frolking, T.A. 1992. A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. *Journal of Geophysical Research*, 97(D9): 9759–9776.

Li, C., Frolking, S. &, Harriss, R. 1994. Modelling carbon biogeochemistry in agricultural soils. *Global Biogeochem Cycles,* 8: 237–254.

Li, C. 1996. The DNDC model. *In:* D.S. Powlson, P. Smith & J.U. Smith, eds. *Evaluation of soil organic matter models using existing, long-term datasets*, pp. 263–267. NATO ASI Series I, Vol. 38. Heidelberg, Springer-Verlag.

Mueller, C.C. 2003. *Expansion and modernization of agriculture in the Cerrado – the case of soybeans in Brazil's center-west.* Department of Economics Working Paper 306, Brasília: University of Brasilia, p. 28.

Nemecek, T., Weiler, K., Plassmann, K., Schnetzer, J., Gaillard, G., Jefferies, D., Garcia-Suarez, T., King, H. &, Milà I Canals L. 2012. Estimation of the variability in global warming potential of worldwide crop production using a modular extrapolation approach. *Journal of Clean Production,* 31:106–17.

Nepstad, D., Schwartzman, S., Bamberger, B., Santilli, M., Ray, D., Sschlesinger, P., Lefebvre, P., Alencar, A., Prinz, E., Fiske, G. & Rolla, A. 2006. Inhibition of Amazon deforestation and fire by parks and indigenous lands. *Conservation Biology*, 20: 65–73.

Pacheco, P. 2012. *Soybean and oil palm expansion in South America: a review of main trends and implications.* CIFOR Working Paper 90. Center for International Forestry Research. Bogor, Indonesia.

Parton, W.J., Schimel, D.S., Cole, C.V. & Ojima D.S. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal,* 51: 1173–1179.

Parton, W.J., Hartman, M.D., Ojima, D.S. & Schimel, D.S. 1998. DAYCENT: its land surface submodel: description and testing. *Global Planet Change*, 19: 35–48.

Paustian, K., Cole, C.V., Sauerbeck, D. & Sampson, N. 1998. CO_2 mitigation by agriculture: an overview. *Climatic Change*, 40: 135–162.

Post, W.M. & Kwon, K.C. 2000. Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology,* 6: 317–327.

Reijnders, L. & Huijbregts, M.A.J. 2008. Biogenic greenhouse gases linked to the life cycles of biodiesel derived from European rapeseed and Brazilian soybeans. *Journal of Cleaner Production,* 16(3): 1943-1948.

Schmidt, G.A., Jungclaus, J.H., Ammann, C.M., Bard, E., Braconnot, P., Crowley, T.J., Delaygue, G., Joos, F., Krivova, N.A., Muscheler, R., Otto-Bliesner, B.L., Pongratz, J., Shindell, D.T., Solanki, S.K., Steinhilber, F. & Vieira, L.E.A. 2011. Climate forcing reconstructions for use in PMIP simulations of the last millennium (v1.0). *Geoscientific Model Development*, 4: 33-45

Soussana, J.F., Tallec, T. & Blanfort, V. 2010. Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. *Animal*, 4(3): 334–350.

Smith, P., Smith, J., Wattenbach, M., Meyer, J., Lindner, M., Zaehle, S., Hiederer, R., Jones, R.J.A., Montanarella, L., Rounsevell, M., Reginster, I. & Kankaanpää, S. 2006. Projected changes in mineral soil carbon of European forests, 1990–2100. *Canadian Journal of Soil Science*, 86: 159–169.

Smith, P., Martino, D., Cai, Z., Gwary, D., Janzen, H., Kumar, P., McCarl, B., Ogle, S., O'Mara, F., Rice, C., Scholes, B. & Sirotenko, O. 2007. Agriculture. *In:* B. Metz, O.R. Davidson, P.R. Bosch, R. Dave & L.A. Meyer, eds. C*limate change 2007: mitigation.* Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). New York, Cambridge University Press.

Sonesson, U., Cederberg, C. & Berglund, M. 2009. *Greenhouse gas emissions in animal feed production decision support for climate certification*. Uppsala, Sweden. Available online: http://www.klimatmarkningen.se/wp-content/uploads/2009/12/2009-2-feed.pdf

Sisti, C.P.J. , dos Santos, H.P., Kohhann, R., Alves, B.J.R., Urquiaga, S., & Boddey R.M. 2004. Change in carbon and nitrogen stocks in soil under 13 years of conventional or zero tillage in southern Brazil. *Soil & Tillage Research:* 76, 39–58.

Vieira, F. C. B., Bayer, C., Zanatta, J.A., Mielniczuk, J. & Six, J. 2009. Building Up Organic Matter in a Subtropical Paleudult under Legume Cover-Crop-Based Rotations. *Soil Science Society of America Journal*, 73(5): 1699–1706.

Wassenaar, T., Gerber, P., Verburg, M., Rosales, M., Ibrahim, M. & Steinfeld, H. 2007. Projecting LUCs in the neotropics: the geography of pasture expansion into forest. *Global Environmental Change,* 17: 86–104.

APPENDIX D

Postfarm emissions

1. POSTFARM EMISSIONS

GHG emissions accounted for in the postfarm part of the food chain include emissions related to fuel combustion and energy use in the transport, processing and refrigeration of products. During this phase of the life cycle, three distinct emission streams were considered: (a) emissions from the transport and distribution of live animals and pork (domestic and international); (b) GHG emissions from processing and refrigeration, and (c) emissions related to the production of packaging material. Excluded from the analysis were estimates of GHG emissions from on-site wastewater treatment facilities, emissions from animal waste[15] at the slaughter site and the consumption part of the food chain (household transport and preparation) and, lastly, disposal of packaging and waste which fall outside the scope of the system boundary studied, but which may warrant further research.

2. ENERGY CONSUMPTION

Energy consumption is the most important source of GHG emissions from the postfarm supply food chain. Table D1 presents regional and country electricity EFs used in this analysis (taken from IEA, 2009). The emission intensity is determined by the mix of fuels used and the efficiency of generation and transmission within a country.

3. EMISSIONS RELATED TO TRANSPORT

The emission intensity of food transport is a function of variables including distance, transport mode, the efficiency of transport loads, the condition of infrastructure (road quality) and factors such as fuel type. The efficiency of different transport modes varies considerably. Air transport has a very high climate change impact per tonnes carried, 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 t-km (tonne-kilometer), compared with estimates of 20–120 g CO_2 per t-km and 80–250 g CO_2 per t-km for rail and road, respectively (Marintek, 2008). Similarly, poor infrastructure, such as bad roads, has an impact on the emission per unit product transported, because it increases fuel consumption. Cederberg *et al.* (2009a) found that because of generally poor road conditions in Brazil, the consumption of diesel there was estimated to be 25 percent higher than under better 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. Table D2 presents average GHG emissions per tonnes of CW transported and demonstrates the impact of load transported (transport capacity) on the average GHG emissions with higher emissions per unit of product transported for lower loads transported.

---

[15] In some countries, manure/slurry from the slaughterhouse is anaerobically digested and the biogas is used for heating and electricity. The central problem is that there is not sufficient 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.

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

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

Source: IEA (2009).

Table D2. Estimated GHG emissions per tonne carcass weight of live pigs transported to slaughter

	Denmark	Sweden	Sweden
Animals transported	280	280	120
Live weight	110	110	110
Vehicle type	Articulated lorry	Articulated lorry	Lorry
Load, tonne	30.8	30.8	13.2
Av. GHG emissions *(kg CO2-eq/tonnes CW/km)*	0.063	0.060	0.220

Source: SIK (2010).

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 product from plant to retail centre. 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, transportation modes, transport EFs and distances.

4. EMISSIONS RELATED TO SLAUGHTER AND PRIMARY PROCESSING OF MEAT

Energy consumption during the slaughter of pigs is used for several processes such as slaughter, evisceration, scalding, singeing, cutting, deboning and also chilling. Average energy use per kg of CW was based on studies from Sweden (Anon, 2002), Denmark, Finland and Spain (Lafargue, 2007) and the EU (Ramirez *et al.* 2006). Due to the limited data on energy use during this phase, in this study we assumed an average value of 2.5 MJ/kg CW and therefore variation in emission intensity has been determined by the average electricity production mix used. 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.

5. EMISSIONS RELATED TO PRODUCTION OF PACKAGING MATERIALS

Packaging is a fundamental element of almost every food product and a key source of environmental burden and waste. The type of packaging used also influences the transport efficiency, since it has its own weight, but also 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, on the other hand, is used to assemble primary packaging to shelter the product during transport and make it possible to transport more of it in one shipment. The climate impact of packaging is one of the least studied aspects within the food chain. Because of the lack of data on the global variations in packaging of meat, this study applies 0.05 kg CO_2-eq per kg CW for both primary and secondary packaging from slaughter-plant to retail.

REFERENCES

Anon. 2002. *Maten och miljön livscykelanalys av sju livsmedel.* (*Food and the environment life cycle assessment of seven food items).* LRF, Sweden.

Cederberg, C., Meyer, D. & Flysjö, A. 2009a. Life cycle inventory of greenhouse gas emissions and use of land and energy in Brazilian beef production. *Swedish Institute for Food and Biotechnology. SIK Report* No. 792.

IEA. 2009. CO_2 emissions from fuel combustion. *IEA Statistics*, Paris: IEA.

Lafargue, P.L. 2007. *LCA of Spanish meals with different protein sources.* Report 2007:3, Department of Energy and Environment, Chalmers University of Technology, Sweden, Gothenburg (MA Thesis).

MARINTEK. 2008. *Updated study on greenhouse gas emissions from ships phase 1 – preliminary results.* Presented to First Intersessional Meeting of the Working Group on GHG Emissions from Ships.

Ramírez, C.A., Patel, M. & Blok, K. 2006. How much energy to process one pound of meat? A comparison of energy use and specific energy consumption in the meat industry of four European countries. *Energy* 31, 12: 2047–2063.

SIK. 2010. *Energy use and GHG emissions from the postfarm livestock chain,* Swedish Institute for Food and Biotechnology.

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 both monogastric species; a general approach is used for both species with a few modifications taking into account differences between production typologies and species.

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

Capital goods including machinery, tools and equipment, buildings such as animal housing, forage and manure storage are 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 and feed and manure storage facilities.

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.

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.

1.1 Animal housing

Three different levels of housing were defined with varying degrees of quality and emissions related to these were calculated (Table E1).

As for hens cages, 2 types were distinguished:

- Conventional cages are usually small enclosures with welded wire mesh sloping floors; they provide equipment only for feeding, drinking, egg collection, manure removal, insertion and removal of hens, and claw shortening. The space is of 430-550 cm^2 per hens;
- Furnished (Enriched) cage FC: in addition to all the equipment found in conventional cages, these cages provide extra elements such as perches, nest boxes, litter area and extra height. The space is of 750 cm^2 per hens.

For pigs, the estimates of embodied energy in materials (steel, concrete and wood) were taken from the Swiss Centre for Life Cycle Inventories database – EcoInvent (300 places). Table E2 provides an example of the life cycle inventory used in the calculation of a high investment structure for fattening pigs.

For laying hens, the estimates of embodied energy in materials were based on the *Euro 2000* group housing system.

Table E1. Typology of animal housing considered in this assessment for pigs and chickens

Level of investment	Characteristics			Production system
	Floor, foundation, walls	**Roof, roof-frame**	**Supports**	
High: *high technology and use of high quality materials*	- slatted floors - material: concrete - multi-tier wire cages (layers)	- material: steel	- stanchions - columns - rafters	- industrial units - peri-urban - more temperate
Average: *intermediate level of technology and use of good quality materials*	- non slatted floors, concrete - walls: local material - wire cages (layers)	- material: steel	- stanchions - columns - rafters	- intermediate - peri-urban - more humid and tropical
Low: *simple housing using local and hand made constructions*	- no walls, floor not paved - scrap wire cages (layers)	- material: steel for pigs, scrap iron for chickens	- material: local	- backyard

Source: Authors.

Table E2. An example of a life cycle inventory for a high investment structure for fattening pigs

Materials	Structure	GWP_{100}* *(kg CO2-eq)*	Quantity of material/unit *(kg of material/ 100 kg LW)*	Emission intensity *(kg CO2-eq/1 kg LW)*
Concrete	Floor	262.61	221.54	1.19
Concrete	Corridors	262.61	39.26	6.69
Concrete	Support - foundation	262.61	25.60	10.26
Steel - structural	Support - stanchions	1.79	4.60	0.39
Steel - structural	Roof frame- rafters	1.79	2.52	0.71
Steel - structural	Roof frame -purlins	1.79	0.42	4.26
Bricks - concrete	Walls	262.61	17.57	14.95
Galvanized metal-shed	Roof	1.79	6.56	0.27
Total				**38.71**

*Global warming potential at 100 years.

Source: Ecoinvent.

The three housing types were then distributed across the different pig production systems (low investment for backyard, average investment for intermediate and high investment for industrial). For chicken commercial systems (broilers and layers), the two first housing systems were considered together.

1.2 Manure storage

The calculation for energy embodied in manure storage facilities was based on a similar methodology and calculation technique outlined above.

For pigs as capital investment, a pit beneath the construction is considered; the dimensions respect those of the EcoInvent pig housing model (300 pig places). The volume of concrete needed to build the pit and the channel to convey the slurry

Table E3. Average emission factors for embedded energy for pigs

Region	Industrial	Intermediate	Backyard
		(kg CO2-eq/100 kg LW)	
OECD	4.75	0.37	0.05
Non OECD LAC	0.46	1.19	0.39
Non OECD Asia	0.63	1.38	0.34
Africa	0.12	0.31	0.59
Non OECD Europe	2.00	1.00	0.23

Source: Authors' calculations.

Table E4. Average emission factors for embedded energy for commercial chickens

Region	Broilers	Layers
	(kg CO2-eq/100 kg LW)	*(kg CO2-eq/100 kg egg)*
OECD	5.89	0.70
Non OECD LAC	1.87	0.20
Non OECD Asia	1.86	0.37
Africa	1.26	0.17
Non OECD Europe	1.61	0.45

Source: Authors' calculations.

pit was calculated. The period of manure storage considered includes 180 days in commercial intensive systems and 90 days in commercial intermediate systems. For chickens, the manure storage facility consists in a 10 cm thick concrete platform and the storage period is 180 days for broilers and layers.

1.3 Feed storage

The calculation for energy embodied in feed storage facilities was based on a similar methodology and allocation technique outlined above. The required feed storage is calculated using an average intake of 4% of body weight for fattening pigs and a total of 286 kg from weaning to finishing (similar values are given in Dalgaard *et al*., 2007). For broilers, an average feed intake of 815 grams per week per bird has been considered. Laying-hens average feed consumption is 629 grams per week for a lifespan of 68 week. As feed storage period is considered the equivalent of the average bird-place per year (5 crops per broilers and 52.5 weeks per laying hens).

Tables E3 and E4 provide averages of emission factors for indirect energy per systems and per regions.

2. DIRECT ON-FARM ENERGY USE

Direct on-farm energy includes the emissions arising from energy use on-farm required for livestock production. Not included is the energy that is used in feed production and transport, as these emissions are included in the feed CO_2 category. Energy is required for a variety of different purposes. Table E5 provides a summary of the main activities for industrial pigs, layers and broilers.

Where more than one type of system was included (e.g. free range, organic) the results for conventional cage systems are used.

Table E5. Main categories of on-farm energy use

	Pigs	Broilers	Layers
Major	Ventilation Lighting Heating	Heating Lighting Ventilation	Ventilation Lighting
Minor	Feeding Manure handling Washing Miscellaneous	Manure handling Feeding	Feeding Manure handling Egg washing/packing Cooling Cleaning Miscellaneous

Source: Horndahl (2008).

2.1 Pigs

Sources for energy use in industrial pig system are given in Table E6.

The most commonly reported direct energy uses are heating, lighting and ventilation. According to Lammers *et al.* (2010) and EC (2003) these account for over 80 percent of the total (though in different proportions; presumably depending on system, climate and weather). The remaining 20 percent is used for a variety of activities, such as washing, housing, lighting, waste handling, feed preparation and delivery.

It was assumed that energy use would be negligible in backyard systems. For intermediate systems, it was assumed that the open-sided housing would mean that there would be no fan ventilation and that heating would only be required for piglets, and that diesel/oil would be the main fuel.

2.2 Chickens

Sources for energy use in layers and broilers system are given in Table E7 and E8.

Note that the results in Table E8 are for secondary energy, i.e. the energy delivered and available for use, rather than primary energy (the total amount of energy consumed, including losses in generation and transmission). Several other studies were available, but were excluded on the grounds that the estimates provided were outliers, or there was ambiguity over the activities included.

Some of the studies, including DEFRA (2007), Wiedemann and McGahan (2011), Leinonen *et al.* (2012a,b), Horndahl (2008) and Nielsen *et al.* (2011), provided breakdowns of the different energy sources. These breakdowns indicated that electricity accounted for 66 percent of the energy used in egg production and 25 percent of the energy used in broiler production.

2.3 Calculating emissions

The average electricity consumption per kg of meat or eggs was multiplied by the EF for electricity in each country, to calculate that country's emissions. The emissions arising from this energy consumption depends on the energy types used and their efficiency of production. In particular, the emissions per unit of electricity vary from country to country depending on types of fuel used, generating plants and transmissions grids.

The non-electrical power sources were assumed to be mainly diesel and the average consumption rate from non-electrical sources was multiplied by a single default EF of 0.0922 kg CO_2/MJ (taken from Berglund *et al.*, 2009).

Table E6. Direct energy use in industrial pig systems

Country	Study	Energy consumption *(MJ/kg LW output)*
Denmark	Dalgaard *et al.* (2007)	0.96
United Kingdom	EC (2003, p110)	1.65
United States of America	Lammers *et al.* (2010)	1.97
United Kingdom	Defra (2007) AC0401	2.93
United Kingdom	Carbon Trust (2006, p24)	1.30
Average		*1.76*
*Electricity**		*1.13*
Other power sources		*0.63*

* Assuming 64 percent of the energy direct use is electricity, based on: Cederberg *et al.* (2009, p18); Defra (2007); Lammers *et al.* (2010); Dalgaard *et al.* (2007); EC (2003, p111).

Table E7. Total direct energy for layers

Country	Study	Energy consumption *(MJ/kg EGG)*
United Kingdom	Leinonen *et al.* (2012a)	2.31
United Kingdom	DEFRA (2007b)	1.57
Australia	Wiedemann and McGahan (2011)	0.88
Sweden	Horndahl (2008)	0.56
Sweden	Sonnesson *et al.* (2008)*	1.14
Sweden	LRF Konsult* (2008, average value)	1.13
Average		*1.26*
*Electricity**		*0.83*
Other power sources		*0.43*

*cited in Sonesson *et al.* (2009).

Table E8. Total direct energy for broilers

Country	Study	Energy consumption *(MJ/kg CW)*
United Kingdom	Leinonen *et al.* (2012a)	6.52
United Kingdom	DEFRA (2007b)	7.50
Australia	Wiedemann and McGahan (2012)*	2.79
Sweden	Horndahl (2008)	4.76
USA	Pelletier (2008)	3.31
Finland	Katajajuuri *et al.* (2008)	5.52
Denmark	Nielsen *et al.* (2011)	1.14
Average		*4.51*
*Electricity**		*1.15*
Other power sources		*3.36*

*average value.

REFERENCES

Berglund, M., Cederberg, C., Clason C. & och Lars Törner, M. H. 2009. Jordbrukets klimatpåverkan – underlag för att beräkna växthusgasutsläpp på gårdsnivå och nulägesanalyser av exempelgårdar. *Delrapport i JoKer-proJeKtet, Hushållningssällskapet Halland.*

Carbon Trust. 2006. *Agriculture and horticulture Introducing energy saving opportunities for farmers and growers CTV009.* Carbon Trust, UK.

Cederberg, C., Sonesson, U., Henriksson, M., Sund, V. & Davis, J. 2009. Greenhouse gas emissions from Swedish production of meat, milk and eggs from 1990 and 2005. *The Swedish Institute for Food and Biotechnology, SIK report* No. 793.

Dalgaard, R. 2007. *The environmental impact of pork production from a life cycle perspective.* University of Aarhus and Aalborg University (Ph.D. thesis).

Defra. 2007. *AC0401: Direct energy use in agriculture: opportunities for reducing fossil fuel inputs.* Department for Environment, Food and Rural Affairs, London.

EC. 2003. *Integrated pollution prevention and control (IPPC)* reference document on best available techniques for intensive rearing of poultry and pigs. European Commission.

Horndahl, T. 2008. Energy use in farm buildings. *Swedish University of Agricultural Sciences Report,* 2008: 8.

Katajajuuri, J.-M. 2007. *Experiences and improvement possibilities - LCA case study of broiler chicken production*. 3rd International Conference on Life Cycle Management, Zurich, Switzerland.

Lammers, P., Honeyman, M., Harmon, J. & Helmers, M. 2010. Energy and carbon inventory of Iowa swine production facilities. *Agricultural Systems,* 103(8): 551–561.

Leinonen, I., Williams, A.G., Wiseman, J., Guy, J. & Kyriazakis, I. 2012a. Predicting the environmental impacts of chicken systems in the United Kingdom through a life-cycle assessment: broiler production systems. *Poultry Sci.,* 91(1): 8–25.

Leinonen, I., Williams, A.G., Wiseman, J., Guy, J. & Kyriazakis, I. 2012b. Predicting the environmental impacts of chicken systems in the United Kingdom through a life-cycle assessment: egg production systems. *Poultry Sci.* 2012 Jan; 91(1): 26–40.

Neuman, L. 2009. Kartläggning av energianvändning på lantbruk 2008. Manuskript. Borås:LRF konsult.

Nielsen, N., Jørgensen, M. & Bahrndorff, S. 2011. *Greenhouse gas emission from the Danish broiler production estimated via LCA methodology*. AgroTech/ Knowledge Centre for Agriculture. Aarhus, Danemark.

Pelletier, N. 2008. Environmental performance in the US broiler poultry sector. *Agricultural Systems,* 98: 67–73.

Sonesson, U., Cederberg, C., Flysjö, A. & Carlsson, B. 2008. *Livscykelanalys av svenska ägg (Life cycle assessment of Swedish eggs).* SIK-report no 783, the Swedish Institute for Food and Biotechnology, Gothenburg.

Sonesson, U., Cederberg C. & Berglund, M. 2009a. *Greenhouse gas emissions in animal feed production decision support for climate certification*. Klimatmarkning För Mat Report 2009: 2

Sonesson, U., Cederberg, C. & Berglund, M. 2009b. *Greenhouse Gas Emissions in Egg Production* Decision Support for Climate Certification. Klimatmarkningen För Mat Report 2009:7.

Wiedemann, S.G. & McGahan, E.J. 2011. Environmental assessment of an egg production supply chain using life cycle assessment. Final project report. *A report for the Australian Egg Corporation Limited.*

Wiedemann, S.G., McGahan, E. & Poad, P. 2012. *Using life cycle assessment to quantify the environmental impact of chicken meat production.* RIRDC publication, No.12/029. Rural Industries Research and Development, Canberra.

APPENDIX F

Allocation to slaughter by-products

The emission intensity depends not just on the amount of emissions and production, but also on how the emissions are attributed to the various products. The results can vary depending on: (a) whether system expansion or allocation is used, (b) which economic or physical properties are used to allocate and (c) which data sets are used for allocation (e.g. which set of prices for meat and other slaughter products).

In certain respects, allocation with pigs is relatively simple compared to some other species. They are kept primarily for meat so, unlike draft animals, there is no need to allocate emissions to labour. As meat production is the predominant function of pig farming, it was decided to allocate all of the emissions to meat. However, it is recognized that some of the emissions could be allocated to the non-meat fraction of the animal, and to the manure produced. Kool *et al.* (2009) identify four distinct categories of pig slaughter by-products:

- fresh meat products and inputs to processed meats;
- food grade fats, rind, bones (and the gelatine derived from rind and bones), organs, entrails and blood;
- feed grade products such as organs, entrails, intestines, bones, head, fat, blood and hair;
- materials that have to be incinerated.

It is estimated that the fresh meat material amounts to 55 percent of the mass and 88 percent of the value of the slaughter products for a conventionally produced Dutch pig (see Table F1).

Wiltshire *et al.* (2009, p58) estimated that the meat from indoor pigs in the UK accounted for 93.6 percent of the economic value of the by-products (including slurry) or 96.9 percent of the slaughter by-products (not including slurry). These results are quite different from those of Kool *et al.* (2009) and illustrate the difficulty of determining the value of by-products in a global study: generalizing from the limited data available may lead to misleading results for some countries. The de-

Table F1. Mass and value of slaughter by-products for a conventional Dutch pig

	Mass *(kg)*	Percentage by mass	Value *(€/kg)*	Percentage by value
Fresh meat	60.7	55	1.85	88
Food and gelatine grade by-products	24.7	23	0.47	9
Category 3 by-products – feed grade	19.5	18	0.21	3
Category 2/1 – energy use	5.1	5	0.00	0

Source: Adapted from Kool *et al.* (2010, p. 27).

Table F2. Emissions intensity of pig meat with and without allocation to slaughter by-products

Industrial - Western Europe	kg CO_2-eq/kg LW	kg CO_2-eq/kg CW
No allocation to by-products	4.6	6.1
Allocation to by-products	4.6	5.4

Source: GLEAM.

cision to allocate all of the emissions to meat was made in light of the lack of global datasets on the value of slaughter by-products. It is recognized that allocating all the emissions to meat will lead to overestimation of the emission intensity in some countries, and future work will seek the data to enable allocation to by-products. Failure to allocate any emissions to slaughter by-products can also lead to double counting if feed ingredients derived from slaughter by-products are included in the ration, and allocated emissions. At the moment, fishmeal is the only animal-derived feed material; however, the rations may be expanded to include other animal-derived feed materials, which will necessitate the allocation of emissions to slaughter by-products.

Table F2 presents an example of results with and without allocation to by-products for industrial pigs in Western Europe.

REFERENCES

Kool, A., Blonk, H., Ponsioen, T., Sukkel, W., Vermeer, H.M., Vries, J.W. & de-Hoste, R. 2009. *Carbon footprints of conventional and organic pork: assessments of typical production systems in the Netherlands, Denmark, England and Germany.* Blonk Milieuadvies, Wageningen University, Wageningen, The Netherlands.

Wiltshire, J., Wynn, S., Clarke, J., Chambers, J., Cottrill, B., Drakes, D., Gittins, J., Nicholson, C., Phillips, K., Thorman, R., Tiffin, D., Walker, O., Tucker, G., Thorn, R., Green, A., Fendler, A., Williams, A. Bellamy, P., Audsley, E., Chatterton, J., Chadwick, D. & Foster, C. 2009. Scenario building to test and inform the development of a BSI method for assessing greenhouse gas emissions from food. *Technical annex to the final report to Defra, Project Reference Number: FO0404.* Department for Environment, Food and Rural Affairs. London.