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LCA METHODOLOGY AND MODELLING CONSIDERATIONS FOR VEGETABLE PRODUCTION AND CONSUMPTION

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1 INTRODUCTION. GOAL AND SCOPE DEFINITION

LLORENÇ MILÀ I CANALS

This report results from a RELU[1]-funded project[2] aiming to validate, from a wide range of disciplines, the advantages or otherwise of eating locally produced vegetables. In other words, it tries to answer the question 'Which is best; to produce vegetables in the UK, or to import produce from overseas?'. To answer this question a range of characteristic vegetables produced in the UK, Spain, Uganda and Kenya are compared considering aspects such as environment, economy, consumer perception, nutrition and community.

The environmental aspects have been assessed applying Life Cycle Assessment (LCA) to a variety of vegetables sourced from different countries. This report explains the LCA methodology followed in the study, as well as the Life Cycle Inventory (LCI) modelling of some life cycle stages, namely those from the retail to the consumption stage. It also explains the LCI considerations of adapting datasets from the ecoinvent database to the requirements of this project. This report is thus mainly a support document for the case studies described in a separate report:

Llorenç Milà i Canals, Almudena Hospido, Ivan Muñoz, Sarah J McLaren (2008): *Life Cycle Assessment (LCA) of Domestic vs. Imported Vegetables. Case studies on broccoli, salad crops and green beans*. CES Working Paper 01/08

[1] The Rural Economy and Land Use Programme (RELU) aims to advance understanding of the challenges faced by rural areas in the UK by funding interdisciplinary research projects (http://www.relu.ac.uk).
[2] Comparative Assessment of Environmental, Community and Nutritional Impacts of Consuming Vegetables Produced Locally and Overseas (http://www.bangor.ac.uk/relu/).

2 LCA IN THE CONTEXT OF THE INTEGRATIVE ASSESSMENT FOR THE ISLE OF ANGLESEY

SARAH J MCLAREN; LLORENÇ MILÀ I CANALS

The overall research question of this project concerns the benefits or otherwise of increasing local production for local consumption of vegetables in the UK. The overall purpose of the LCA studies is the investigation of the environmental impacts associated with different systems for vegetable production, in order to inform the effects of increasing local production for local consumption of vegetables. More specific objectives include:

1. Determining which life cycle stages of selected vegetables contribute the greatest environmental impacts.
2. Comparing UK and overseas production of selected vegetables that are consumed in the UK.
3. Investigating whether differences in production practices between farms are more significant than differences between countries.
4. Analysing the impacts of a change towards more local production for consumption on the island of Anglesey.

The first three goals can be directly addressed with the LCA results (as done in Milà i Canals *et al.* 2007a). The scope of the LCA studies for this RELU project includes the assessment of vegetable production and delivery to UK consumers, as well as food storage, preparation and consumption at home. Different levels of detail will be required for the data collected, according to the goals of the study, with site-specific data for the studied farms, national statistics for food retail and literature data for the production of ancillary products (fertilisers; pesticides; fuels; farm machinery; electricity; etc.). However, the 4th goal mentioned above requires the integration of the LCA results with the other aspects assessed in the project, which needs to be considered carefully.
For the first three objectives, analysis from a retrospective perspective is relevant. In other words, data on current activities can be collected and analysed. However, for the fourth objective a prospective perspective is appropriate. In this case, several additional factors must be taken into consideration in constructing the conceptual model. Figure 2-1 shows the current situation for Anglesey. It can be seen that a relatively small proportion of land is currently used for horticultural production.

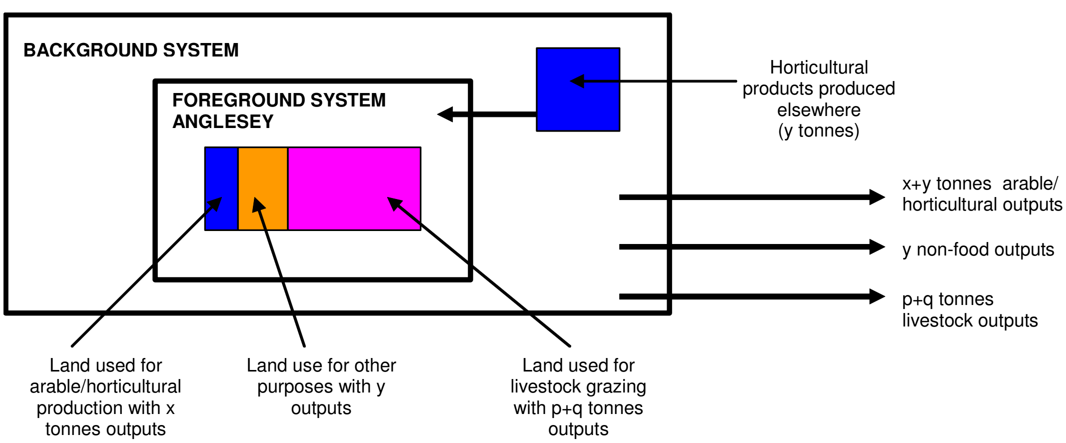

Figure 2-1: Current Situation on Anglesey

Figure 2-2 shows the future scenario for Anglesey with more local production of horticultural crops. This assumes that horticultural production has expanded on Anglesey at the expense of livestock production which has been displaced elsewhere. In order to analyse the impacts of more local production of horticultural crops, the consequences of displacing livestock production must be taken into account. In fact, the analysis becomes:

Increased horticultural production on Anglesey and local onward transport
Minus horticultural production elsewhere and its onward transport
Minus displaced livestock production on Anglesey and its onward transport
Plus replacement livestock production elsewhere and its onward transport

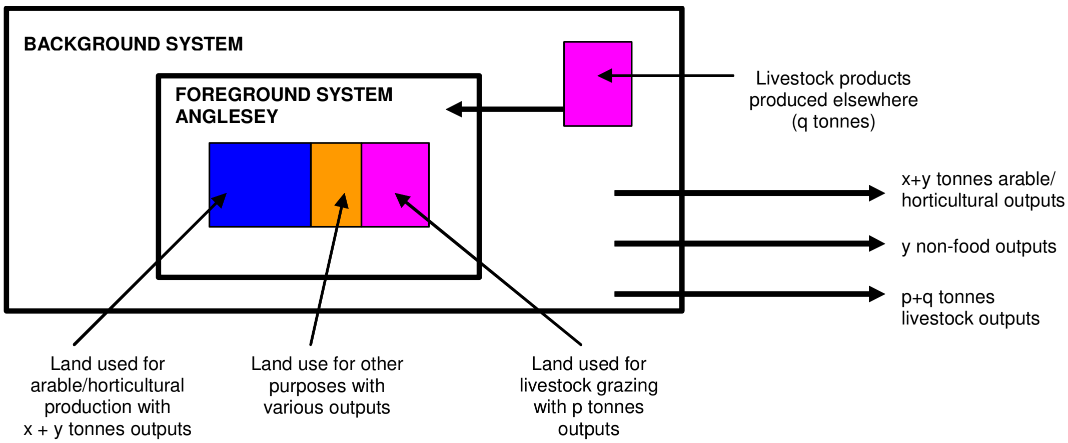

Figure 2-2: Future Scenario for Anglesey

If it is assumed that yields and environmental impacts remain the same wherever the agriculture takes place, then the only relevant changes to be modelled are the differences in transport distances between points of production and consumption. If the changes in location of production lead to changes in yields and environmental impacts due to different production practices, then the net changes must be included in the analysis. For example, if future additional horticultural crops on Anglesey have lower yields and higher environmental impacts compared with current production elsewhere that is being displaced, then the net increase in environmental impacts should be modelled. However, if, at the same time, the displaced livestock production on Anglesey leads to more efficient livestock production with lower environmental impacts elsewhere, then this could cancel out the net increase in environmental impact associated with the additional horticultural crops on Anglesey.

It can be concluded that, if yields and production practices are similar in different areas, then the main changes in environmental impacts will be associated with changes in transport distances between points of production and consumption – and in general local horticultural production on Anglesey will be beneficial (from an environmental perspective). If relatively higher yields and better production practices mean less environmental impacts for horticultural production on Anglesey, and less environmental impacts for livestock production elsewhere, then local horticultural production will be beneficial. If relatively lower yields and worse production practices mean higher environmental impacts for horticultural production on Anglesey, and/or greater environmental impacts for livestock production elsewhere, then it is unclear whether local horticultural production will be beneficial. In fact, there will be a trade-off between a net increase in agricultural production impacts and a net decrease in transport impacts.

It can be seen that the objectives of the analysis require two types of analysis to be undertaken in the study:

1. Hot-spot analysis: assessment of current life cycles of selected food items.
2. Comparative analysis focused on:

i. Assessment of consuming selected food items on Anglesey that may have been be produced in different countries and/or on different farms.

ii. Assessment of consuming different – but substitutable – food items on Anglesey at different points in the year.
iii. Assessment of future horticultural production of selected crops on Anglesey compared with current production of selected crops elsewhere that will be displaced, AND assessment of future livestock production elsewhere compared with current livestock production on Anglesey that will be displaced.

3 LIFE CYCLE INVENTORY (LCI) MODELLING FOR ON-FARM OPERATIONS

LLORENÇ MILÀ I CANALS

3.1 Production and maintenance of farm machinery

It is commonly suggested in agricultural LCA that the production of machinery and other capital equipment should be included in the inventory because they can have a relevant share of the overall impacts (Audsley *et al*. 1997).

According to the project scoping, site-specific data have been collected from farms in the UK, Spain, Uganda and Kenya, while more generic data have been used for upstream production of farm inputs and downstream activities. Site specific data on machinery use (use per year, expected lifetime, weight, etc.) have been collected from the studied farms in order to allocate the impacts of machinery production to the studied crops. As for farm inputs production, including machinery, the ecoinvent database has been used throughout the project to keep consistency. The method suggested in Audsley *et al*. (1997) is generally followed in the ecoinvent database (Nemecek *et al*. 2004), where it has been implemented with a more sophisticated model (specific study of machinery production related emissions; detailed materials composition; etc.). The assumptions and data conversions for the different life cycle stages of machinery considered in this study are explained in the following sections.

3.1.1 Manufacture

Energy consumption and materials composition are representative of different agricultural machines, and have therefore been used as they appear in ecoinvent. Specific emissions from manufacture are included in ecoinvent: NMVOC from solvents and fuel and CO_2 from varnish corrosion (insignificant in the overall life cycle). However, the reference flow for machinery datasets is 1 kg of machine, and this has been changed to hours (for tractors and other self-propelled machines) or hectares (for tillage machines) to reflect the data collected in the inventory. When doing so, site-specific data on machinery weight, lifespan and yearly usage have been used to parameterise the ecoinvent data (Nemecek *et al*. 2004, p. 52) in the following way:

$$\left(\frac{env.\,flow}{kg\ machine}\right)_{ecoinv} \times \frac{(weight\ machine)_{RELU}}{(total\ units\ usage\ in\ machine's\ lifetime)_{RELU}}$$

where the first element represents the flows recorded in the ecoinvent datasets (referred to 1 kg of machine) and the second element is the parameter “kg_ha” that renders the ecoinvent data more representative of the data collected in our study; the sub index ‘RELU’ refers to data collected specifically in each farm. The allocation to the total units (hours or hectares) used in the machine’s lifetime is done in the ecoinvent datasets for field work processes, and thus needs to be removed from there once it has been done in the machine’s manufacture.

3.1.2 Transport of machinery

The ecoinvent database considers machinery transportation within Western Europe totalling 500km in train and truck. Many machines used in the UK are actually produced in the UK, which might lead to lower transportation distances. However, this would be balanced by longer distances for machines imported from the continent. In any case, transportation of machinery is likely to be irrelevant, and so no changes have been considered for the studies in UK and Spain. A special consideration should be done for the studies in Kenya and Uganda, where machinery might be manufactured in distant countries.

3.1.3 Maintenance and repairs

The considerations done in ecoinvent for maintenance (change of tyres, mineral oil, filters, batteries, etc.) are considered valid for this project. In the case of repairs, an increase of the manufacture materials is considered depending on the machine type (Nemecek *et al*. 2004, p. 49). For tillage machines this is considered to be 45% extra material (steel); as specific data on this materials is easily collected in the farms (representing the frequency of change of tillage components such as harrow tines), this will be used instead. Therefore, the steel input in the ecoinvent datasets for tillage machines is reduced by 45% and then increased by the calculated site-specific amount. The data collected from farmers actually shows quite dramatic increases in steel consumption when calculated like this, with e.g. increases of 200-264% (instead of the suggested 45%) for repairs in ploughs and power harrows. The following steps are done for the parameterisation:

1. The "factor" in the main steel entry is divided by 1.45 to obtain the amount of steel actually used in the machine's manufacture.
2. a new parameter is created: total_steel_ha = kg_ha + ka_ha_spares
3. total_steel_ha is used as alias for the main steel entry

The proportional use of a farm building (shed + garage) is allocated to farm machinery in ecoinvent. However, the data provided by ecoinvent is representative of specific building types in Switzerland, where buildings tend to be more expensive and solid than in other countries. Therefore, even though the impacts from farm buildings may be relevant from an environmental point of view in LCA, they are NOT considered in the present study, because the uncertainties included with them would probably be as high as their values. However, in order to answer one of the many research questions addressed in this project, the land occupation associated to farm buildings for the storage of machines has been included. The data on land use have been obtained from ecoinvent (Nemecek *et al*. 2004).

3.1.4 Land use associated to farm buildings

Nemecek *et al*. (2004, table A.10) offer data on space requirements for different machines. It has been assumed that a shed is available in all farms to shelter all machines, and that a space equivalent to the requirement of each machine is provided all year-long. Therefore, the data in m^2 offered by ecoinvent (see above) are directly converted to m^2year for each machine. The m^2year are then allocated to the functional output of the machine during one year. Area occupied by farm sheds is classified as 'Occupation, urban, discontinuously built' in ecoinvent.

A similar approach has been used for the other buildings in the farm used for the studied vegetables: greenhouses for plant propagation and potato chitting, stores, packing plants, etc. The area used by these buildings has been obtained from the farmers and classified as 'Occupation, urban, discontinuously built'.

Specific data for land use by farm buildings are provided in LCA reports for the different farms studied.

3.2 Use of agricultural machinery (field works)

Fuel consumption for the different operations has been assessed specifically for the studied farms. This figure has then substituted the figures reported in ecoinvent, plus all subsequent emissions related to fuel consumption. The same sources used in ecoinvent for fuel emissions in agricultural machinery have been used, specifically for CO, HC (expressed as NMVOC) and NO_x (Nemecek *et al*. 2004, Table A10), which differ substantially respect road vehicles. The emissions of CO, HC, NO_x are expressed in g/h (Nemecek *et al*. 2004, Table A10), depending on each different operation; these emissions are re-calculated with the duration of the operations obtained from the farmers using the parameter rate_h (dividing the duration in hours/ha obtained from the farmers by the duraction expressed in ecoinvent (Nemecek *et al*. 2004, Table A9). To update fuel-related emissions (CO_2, SO_2, Pb, methane… Nemecek *et al*. 2004, table 7.1) the parameter rate_fuel (fuel consumption per ha in RELU divided by fuel consumption per hectare in ecoinvent) is created and used for multiplying inputs (fuel consumption) and outputs related to fuel (most air emissions).

The emissions to soil from tyre abrasion are calculated from replacement of tyres and an estimate of the amount of rubber rubbed off; these emissions have been considered as they are in ecoinvent.
The checks are reported as comments in the Data quality –Technique (this has to be done every time the process is used):

- Completely representative: duration of operation lies within ±20% of that reported in ecoinvent
- Partly representative: duration of operation lies within ±21-50% of that reported in ecoinvent
- Not representative: duration of operation is over ±51% different of that reported in ecoinvent

3.3 Consideration of manual labour

With very few exceptions (e.g. Piringer and Steinberg 2006; Nguyen and Gheewala in press) the environmental impacts associated with human labour have systematically been excluded from LCA studies. The reason most often argued for this[3] is that labour-force maintenance-related environmental impacts (e.g. food consumption by workers; energy use for shelter; etc.) would occur regardless of the studied system (Piringer and Steinberg 2006). I.e. that person would still eat (and possibly work elsewhere) if the studied system was not in place. Piringer and Steinberg (2006) assess the energy costs of labour in wheat production in the USA, concluding that this is of minor importance. According to their findings, labour-related energy would represent maximum 7.1% of energy use for wheat if the highest estimate for labour energy use is compared to the best estimates (i.e. not highest values) for the other items of the energy bill. It should be noted that there is a huge uncertainty in this value. In any case, it could be argued that 'in terms of energy efficiency at least, it would be a little unfair to compare the energy balance of non mechanised or partly mechanised systems with fully mechanised ones without accounting for human labour input' (Shabbir Gheewala, 19.06.2007 e-mail communication in LCA forum).

In this study we have considered that impacts of maintaining humans are not affected by the studied system (i.e. food consumption, housing, etc. are excluded from the study), but that work-related transportation is increased by the studied system. Hence, an estimation of labour related transport has been done for labour-intensive operations.

The nature of labour force in agricultural sector varies widely between the assessed countries, and so the way in which these impacts have been assessed also varies. In any case, the attempts done in this study have to be seen only as a first try to assess the relevance of labour transport-related impacts, and not as an exhaustive absolute statement of environmental impacts related to agricultural human labour in different countries.

3.3.1 'Labour-intensive' operations

First of all, a focus has been placed on those operations that the farmers consider as 'labour intensive'. These are generally all operations that cannot be mechanised, such as harvesting of lettuces, brassica or green beans; hand weeding within rows; installation/removal of irrigation infrastructure; etc. In the UK and Spain most of these operations coincide (with a trend in Spain to perform more operations manually), whereas in Uganda the assessed farms show a much lower degree of mechanisation, with use of tractors and machinery being the exception rather than the rule. However, in Uganda most farm workers travel to the field by bike or on foot, and so their transportation impacts have been neglected.

The labour intensive operations recorded for the LCA studies do not match the labour costs that could be found in the farm accountancy books. As a rule of thumb, all permanent workers would be omitted from the LCA study, because they generally perform operations with high energy use (e.g. mechanised farm operations, where the tractor fuel use will override the fuel use of their private cars) or with low labour input per unit of product (e.g. in a packing plant). On the other hand, it is usually the temporary workers who perform the labour-intensive operations. This study has tried to provide a first estimate of the importance of transportation of temporary workers for some of the studied crops.

[3] E.g. In June and July 2007 there was a long on-line discussion in the international LCA discussion forum; contributions by the author, Shabbir Gheewala, Gabor Doka, Rolf Frischknecht, and Gherard Piringer have been used in this section.

3.3.2 Plane transportation of immigrant workers

The UK farms show a particular pattern in terms of hand labour. Without attempting to provide a sociological picture of farm workers in the UK, there seems to be less agriculture-related permanent immigration than in e.g. Spain. Indeed, some of the studied UK farms participate in schemes such as SAW: Student Agricultural Workers. The idea of such schemes is to bring in young workers from abroad, usually Eastern European countries, to work in farms only for the most labour-demanding seasons. These workers (usually students, and thus the name of the scheme) return to their home countries after the growing season. SAW are usually lodged in on-farm facilities, and so their transportation requirements during the growing season are often limited within the farm. These have been quantified in four of the participating farms.

Potentially more relevant than the on-farm transportation is the transportation related to “importing” hand labour to the UK and then returning it to the home countries. This is usually done by plane and has been quantified wherever possible. E.g. a farm brings ca. 400 SAW to work on the growing season of green beans and asparagus, representing a total of 370ha and resulting in 1.08 SAW per ha per crop. Most of these workers come from Eastern Europe (mainly Poland, Bulgaria and Ukraine) and some from South Africa. For the sake of simplicity all of them have been considered to come from Poland (main country of origin), and a round plane trip of 3,260km (Warsaw-UK-Warsaw) per SAW has been considered, or **3,520 passenger kilometre per ha per crop**.

3.3.3 Road transportation of locally resident workers

In Spain, most of the temporary workers are permanent immigrants from Northern or Sub-Saharan Africa and South America. Such workers live permanently in Spain and move from region to region to follow growing seasons, although they are mostly based in agricultural counties such as Murcia, Almeria, Lleida, etc. Therefore it is not justified to include their transportation from their home countries, but only the road transportation they have everyday to the farms. To do so, apart from the labour intensiveness of different operations the average transportation distances and means of transportation have been recorded. E.g. most farms recruit their workers from neighbouring villages, and the workers often travel in vans of 8-9 people doing daily return journeys of ca. 30-40km. This type of transportation is also relevant for UK farms, although when SAW are considered they often use buses for the group transportation of workers within the farm.

3.4 Soil emissions from fertilisers

Data on fertiliser production used for the LCA have been obtained from an existing study (Davis and Haglund 1999, used within the ecoinvent database), as well as their application in the field (as described by the farmer). Nutrient-related emissions from soil measured and modelled for this RELU project are included in this section (NH_3; N_2O; NO_x; NO_3^-; CH_4); default literature values have been used while compiling the field measurements:

- NH_3-N emission factors (expressed as % loss of N content) from Asman (1992) have been used following the recommendation of Audsley *et al*. (1997, p. 42); see Table 3-1.
- For the calculation of N_2O emissions, the emission factors for mineral fertilisers (Armstrong-Brown *et al*. 1994) have been used; see Table 3-2. For organic fertilisers, the content of nitrate and ammonium N has been used with the factors in Table 4 for nitrate and ammonium.
- NO_x-N has been considered as 10% of N_2O-N (Audsley *et al*. 1997, p.49).
- NO_3^- and PO_4^{3-} have been obtained from literature values as 15 kg N-NO_3^- $ha^{-1}year^{-1}$ and 1kg P-PO_4^{3-} $ha^{-1}year^{-1}$ (Cowell 1998).
- An emission of 1 kg of CH_4 to the air per each 150 kg of N applied as ammonium fertiliser has been considered (Audsley *et al*. 1997, p.58).
- Whenever animal manure is used, 20% of its N content is assumed to be in ammonium form, 30% in urea form, and 50% in non soluble organic form (disregarded for emissions calculations)

Table 3-1: Emissions of Ammonia (NH_3-N as % loss of N content) from mineral fertilisers

INPUTS (Mineral fertilisers)	Ammonia (NH_3-N) emissions to air (% loss of N content)
Ammonia, direct application	1
Ammonium nitrate	2
Ammonium phosphate	4
Ammonium sulphate	8
Calcium ammonim nitrate	2
Compound N	4
Nitrogen solutions	2.5
NK N	2
NPK N[a]	4
Other NP N	3
Other straight nitrogen	2.5
Total straight nitrogen [b]	4
Urea	15

[a] Assumed to be half nitrate, half ammonium
[b] This should only be used if no information is available on fertiliser consumption of the individual categories

Table 3-2: Emissions of Nitrous Oxide (N_2O-N as % loss of N content) from mineral fertilisers

INPUTS (Mineral fertilisers)	Nitrous Oxide (N_2O-N) emissions to air (% loss of N content)
Ammonium (soil temperatures 0-10ºC)	0.4
Ammonium (soil temperatures 10-20ºC)	0.5
Nitrate (soil temperatures 0-10ºC)	1.7
Nitrate (soil temperatures 10-20ºC)	1.1
NPK N[a] (soil temperatures 0-10ºC)	1.05
NPK N[a] (soil temperatures 10-20ºC)	0.8
Urea (soil temperatures 0-10ºC)	0.8
Urea (soil temperatures 10-20ºC)	3

[a]Assumed to be half nitrate, half ammonium
Source: Adapted from Armstrong Brown *et al*. (1994) in Audsley *et al*. (1997)

3.5 Field carbon emissions

The treatment of carbon emissions in LCA of biotic production systems has generated much controversy and been treated inconsistently by different practitioners for many years. Possible reasons for this include lack of inventory data for emissions from agro-forestry ecosystems, as well as different perspectives for different system boundaries: the relevance of C fixation through photosynthesis seems to be perceived differently when the waste treatment stage of the bio-based product (when usually all C is re-released through aerobic or anaerobic degradation) is included. Most bio-based LCA studies are limited to the cradle-to-gate stages.

Three basic approaches to the treatment of biogenic C in LCA studies may be distinguished:

1. Do not consider CO_2 fixation by vegetation and neglect the downstream biogenic CO_2 emissions.
2. Consider CO_2 fixation by vegetation as a negative emission and then account for the emission wherever it occurs (e.g. in waste treatment).
3. Perform a full carbon balance of the agro-ecosystem and account for all subsequent emissions in their relevant form.

Option 1 has often been preferred, possibly because C fixation is difficult to measure. However, this approach presents some inconsistencies; e.g. if the C fixed as CO_2 is emitted as CH_4 (e.g. from enteric fermentation in livestock production, or from biomass fermentation in landfills) then it is recommended to include the emissions even if the C is biogenic. This option has been used e.g. in Cederberg and Mattsson (2000); Milà i Canals *et al*. (2002; 2006); Herrmann *et al*. (2007).
Option 2 is preferable in terms of consistency and completeness (Rabl *et al*. 2007), but it presents many challenges and has been followed in LCA studies in varying degrees of sophistication. In fact, to be fair not only fixation in biomass but also emissions by agro-ecosystems should be assessed; i.e. an ecosystem carbon balance is required. Yet, most LCA studies trying to account for biogenic C emissions are limited to assess the amount of C fixed in the harvested plant tissues (e.g. the ecoinvent database: Jungmeier *et al*. (2003); Nemecek *et al*. 2004; Nebel *et al*. 2006) and its subsequent release in the use or waste stages. Some of the complexities required for a full carbon balance approach, and not often considered in LCA studies include:

a. C fixation does not only happen in the plant harvested tissues, but also in non-harvested biomass, roots and soil organic carbon: SOC (roots, decomposition + synthesis to SOC, etc.). The effects of land use practices on SOC may be of the same order of magnitude as emissions from fuel combustion according to IPCC (2001), and this has been systematically omitted from LCA studies up to date. Notable exceptions include the following: Kim and Dale (2005) use the Century SOC model to predict changes due to different tillage practices in the production of bio-polymers; Milà i Canals *et al*. (2007c) review the available methods to include changes in SOC caused by management practices in LCA studies; Brandão *et al*. (submitted) use literature values to assess SOC changes under different bio-energy crops in the UK.
b. When agro-ecosystems act as a net sink of C, it needs to be recognised that C stored in biomass might be re-released in relatively short periods of time (years or decades). The C storage time in SOC and biomass should thus be factored in the assessment of the beneficial effects on GWP, as suggested by Nebel and Cowell (2003).
c. In order to consider a full C balance, one needs to quantify all downstream C emissions, including those that seem awkward from a LCA perspective such as CO_2 due to human respiration. This has not been contemplated in LCA, although for food LCA this is the natural "use phase" (in the same way that CO_2 emissions from combustion need to be included in the assessment of biomass for energy). Subsequent emissions include the human wastewater treatment and emissions related to faeces and urine excretion. No models to include these emissions have been available until now; Muñoz *et al*. (2007; submitted) suggest a first attempt.

In ecology and atmospheric sciences the net release/uptake of C by ecosystems has been the subject of research for many years. Chapin *et al*. (2006) review the main approaches and concepts related to the estimation of the Net Ecosystem Carbon Balance (NECB, a new term suggested by Chapin and co-workers to define the net rate of C accumulation in –or loss from- ecosystems). The main concept used traditionally by ecologists and soil scientists to approach NECB is the NEP (Net Ecosystem Production), which is a good approximation to NECB for short time scales and when there is little transfer of dissolved C into or out of the system. NEP is thus adequate to estimate the NECB of agricultural systems. Koerber *et al*. (forthcoming) address the importance of properly assessing such concepts in agricultural LCA, and describe the experimental methods to quantify NEP.

For this project, NEP has been measured at field scale. As an interim approach literature values have been used to quantify the most important parameters in the calculation of NEP. As explained by Koerber *et al*. (forthcoming), NEP may be calculated as the balance between NPP (Net Primary Production, equivalent to all photosynthetic fixation minus autotrophic respiration) and R_{soil} (soil respiration, also called heterotrophic respiration). As a first approximation to these parameters, NPP has been assumed to equal the C content in harvested biomass (i.e. neglecting C fixed in non-harvested plant biomass: crop residues including roots), and R_{soil} has been estimated from the change in SOC, ΔSOC (i.e. neglecting heterotrophic respiration of plant residues). The rationale behind this is that literature values are available for these parameters and to some degree the fixation not accounted in the NPP is balanced by the emissions not considered within R_{soil}. Table 3-3 provides values for C content of the crops studied here, and Table 3-4 shows the values considered for ΔSOC. Particularly for the latter, the variability of the values is enormous, and the results thus obtained should only be seen as a first rough approximation.

Table 3-3: C contents in harvested plant tissues for the different crops (kg C/kg crop).

Crop	*C content [kg C/kg crop]*	*Source, comment*
Lettuce	0.034	From own field measurements in 3 countries
Broccoli	0.046	Calculated from raw broccoli composition (Food Standards Agency 2002) and excretion model (Muñoz *et al.* 2007)
Green beans	0.036	Calculated from raw French beans composition (Food Standards Agency 2002) and excretion model (Muñoz *et al.* 2007)
Peas (shelled)	0.112	Calculated from raw shelled peas composition (Food Standards Agency 2002) and excretion model (Muñoz *et al.* 2007)

Table 3-4: Literature values of ΔSOC for the European and African farms used in this project (a negative sign indicates loss of SOC; a positive sign indicates build-up of SOC).

	ΔSOC [t C $ha^{-1}year^{-1}$]	*Source*	*Comments from source*
Vegetable cropping, Europe	-0.4	Arrouays *et al*. (2002, p. 138)	Annual crops in France (review of several studies)
Vegetable cropping, Africa	-0.9	Woomer *et al*. (1997)	Continuous cultivation in Kenya

The 4.62% of C in fresh broccoli (Table 3-3) is translated into 169.51 kg CO_2 per tonne of fresh produce harvested. This has been implemented in the GaBi process “crop, cradle to farm gate” as can be seen in Figure 3-1: the yield per ha is inserted as a free parameter in tonnes per ha, and this parameter is transformed by the factor of kg CO_2 per tonne with a negative sign to denote a fixation (i.e. “negative emission”). SOC-related emissions have been included in the soil management process within GaBi (see Figure 3-2). The figure shows the case for broccoli; as two crops per hectare have been considered, the 400kg of C emitted per year have been divided amongst the two crops (i.e. 733 kg CO_2 per crop).

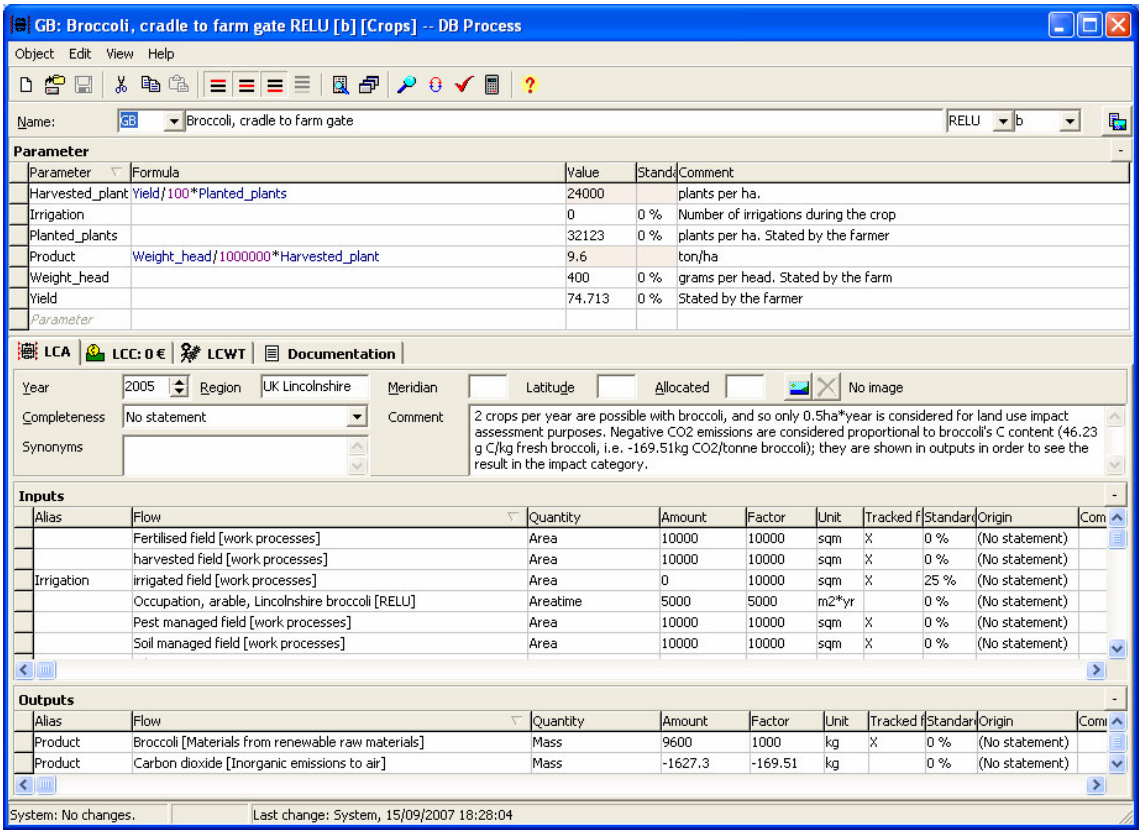

Figure 3-1: Consideration of carbon fixation in crop biomass in the GaBi process "Crop, cradle to farm gate".

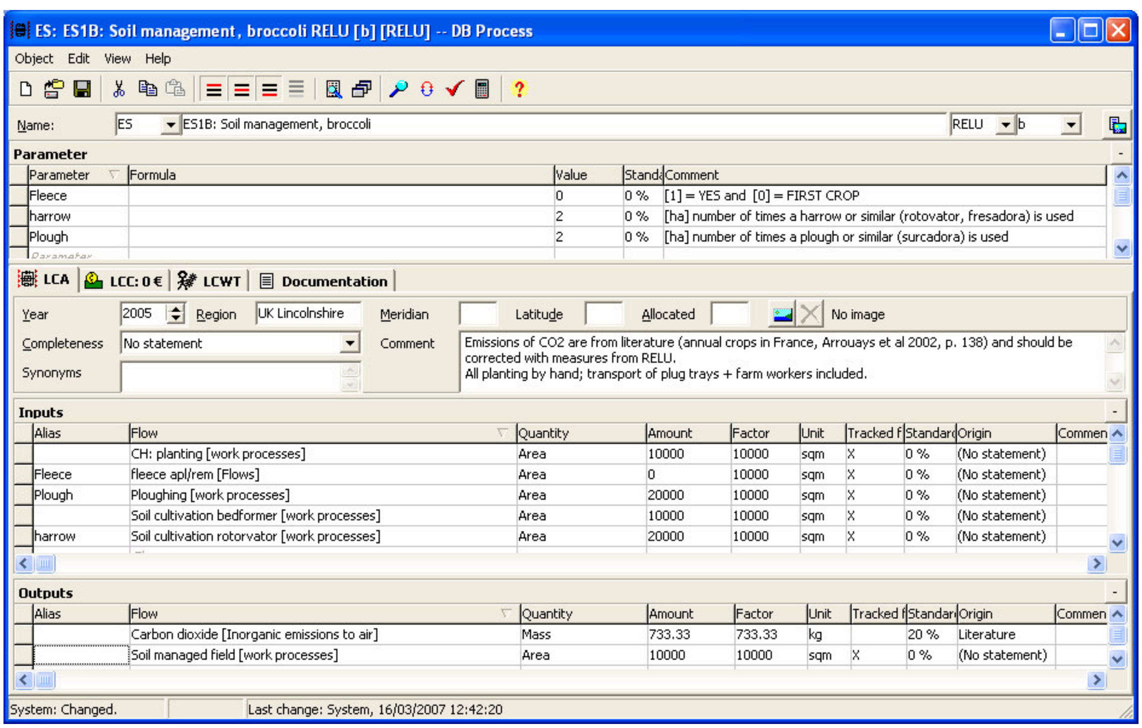

Figure 3-2: Consideration of emissions from SOC degradation in the GaBi process "soil management".

Estimating these two components of NEP is, to our knowledge, the best attempt to date in an LCA study to match the ecosystem C emissions in a way compatible to ecology modelling. The first results are published in Muñoz *et al*. (submitted).

In parallel, this project offers for the first time field measurements of NEP to compare with the literature estimates. Measured NEP values may be used to substitute the fixed C in biomass and the SOC degradation values, by including a parameter "NEP" in the "crop, cradle to gate" process and a flow for CO_2 emission with a negative factor. The factor needs to express the conversion factor from NEP units (tonnes C) to CO_2 (i.e. 44 kg CO_2 per 12 kg C, or a factor 3.6667). This is shown in Figure 3-3 with a fictitious value for broccoli cropping, where the field is a net emitter of 250 kg C per ha per year. The annual NEP value is typed in the free parameters area, together with the number of crops produced per year; the NEP allocated per crop is then used to quantify the CO_2 emissions with a negative factor of -3.6667. If the NEP typed in is a positive value, indicating a net gain by the agro-ecosystem (i.e. the ecosystem is a net sink), then the emissions are negative.

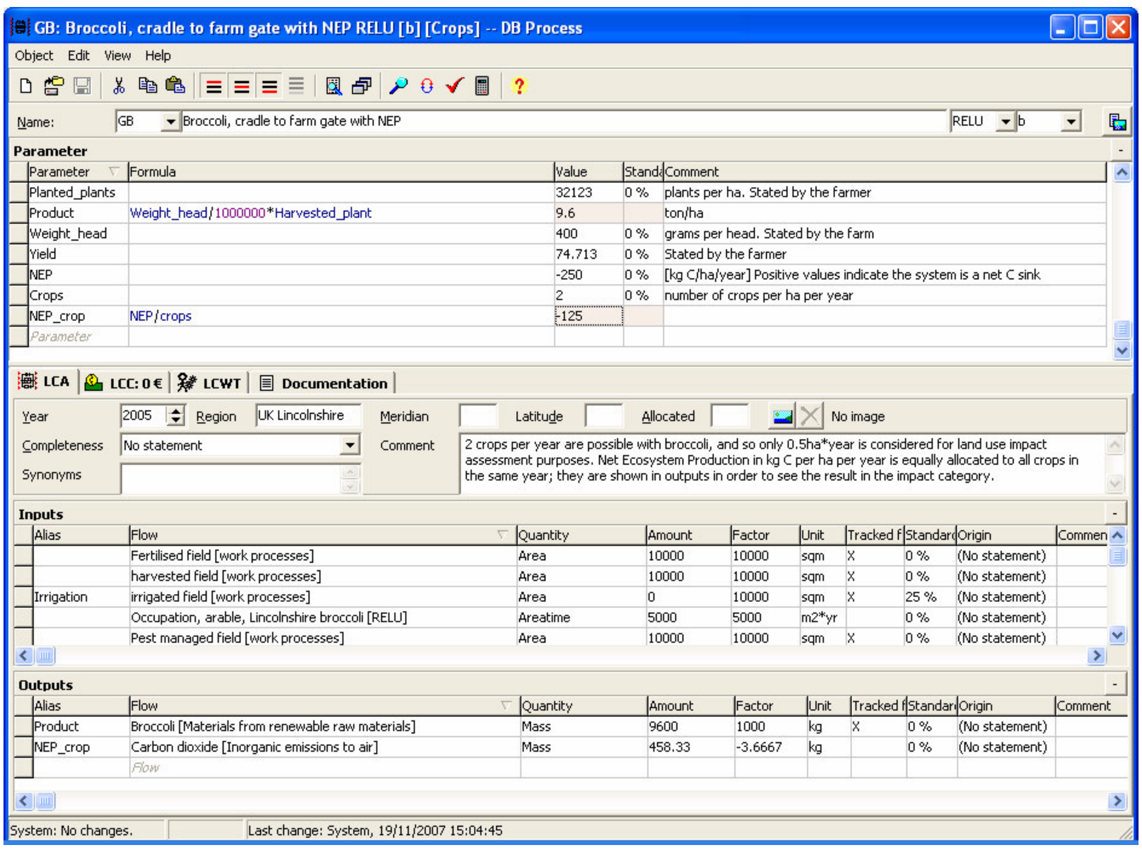

Figure 3-3: Consideration of NEP values in the GaBi process "Crop, cradle to farm gate".

4 LCI MODELLING FOR TRANSPORTATION AND POWER GENERATION IN AFRICAN DATASETS

LLORENÇ MILÀ I CANALS

Extensive transportation data, including production, maintenance and disposal of infrastructure (e.g. roads and vehicle fleet) are available in ecoinvent (Spielmann *et al*. 2004) for the main transportation systems (road, rail, air, water). Extensive data are also available for energy delivery systems, including power. These have been used as the reference datasets for this project. However, no information on electricity generation in Africa is offered in Ecoinvent, and so specific datasets have been developed in this project as explained in this section.

4.1 Manufacture

Contrary to what is suggested in ecoinvent, manufacture of transportation infrastructure (vehicles and roads) is not considered in this study. This consideration follows normal practice in LCA, where production of capital goods is not included unless it is expected to cause significant impacts (as in the case of agricultural machinery). Vehicles and other infrastructures (e.g. roads) are used quite intensively, and therefore they are not included.
However, in order to answer one of the many research questions addressed in this project, the land occupation associated to road (and other infrastructures, such as airports and food processing plants) has been included. The data on land use have been obtained from ecoinvent (Spielmann *et al*. 2004).

4.2 Land use associated to road transportation

Spielmann *et al*. (2004) offer statistical data to allocate road use to the service of goods transportation (expressed in ton*kilometres: tkm) and passenger transportation (expressed in passenger*kilometres: pkm) in Europe. It must be noted that they have allocated road use data based on vehicle kilometric performance (number of kilometres run by any type of vehicles); this is considered to be fair for the land use flow. If gross transport performance (based on tkm transported) or axle-weight were used as a basis for allocation, the results would change drastically (Spielmann *et al*. 2004, p.92). The following values, representing the European road system, have been used in this project:

Table 4-1: Specific land use and road operation in Europe.

	Van	Lorry 16t	Lorry 32t	Passenger car	Bus and coaches
Road demand	2.83E-03 m*year/tkm	6.73-04 m*year/tkm	1.61-04 m*year/tkm	7.11-04 m*year/pkm	7.07-05 m*year/pkm

These factors need to be combined with the land use values per m of road; Spielmann *et al*. (2004) only offer such values for Switzerland, and suggest they can be considered as valid for other European countries. We consider only the land occupation, as trends of land transformation from any type of land use to road vary immensely from year to year. Values of 6.43m^2 of road network plus 1.36m^2 of road embankment are considered per m of average Swiss road are considered in this project (Spielmann *et al*. 2004, p. 103).

It should be noted that African roads (at least the ones seen in Uganda and Kenya) are by no means similar to standard roads in Europe; both their dimensions and use intensity differ greatly. However, the abovementioned factors for land use have been considered in this study due to lack of more relevant data.

4.3 Operation of road vehicles

The ecoinvent database offers datasets for the operation of empty and full trucks (16t; 28t and 40t trucks).

4.3.1 Operation of conventional trucks

The fuel consumption in kg fuel per km expressed in ecoinvent datasets has been used to construct new, parameterised, datasets, where the user may introduce the distance (in km) and average payload carried (in tons) to calculate final impacts together with the amount of product considered in the study (entering and leaving as a 'cargo' flow). The user may also change the diesel consumption for the empty and full trucks (in litres/km) to adapt to specific fleet efficiency, or leave the default values (from ecoinvent).

The parameters used in the datasets are thus:

Free parameters (*name*-[units])	**Fixed parameters (*name*: formula-[units])**
Avg_cargo-[t]	*diesel_dens*: 0.845-[kg/l]
cons_empty-[l/km]	*cons_empty_kg*: cons_empty*diesel_dens-[kg diesel/km]
cons_full-[l/km]	*cons_full_kg*: cons_full*diesel_dens-[kg diesel/km]
Distance-[km]	*tot_cons*: (cons_empty_kg+(cons_full_kg-cons_empty_kg)/28*Avg_cargo)*Distance-[kg diesel]
	Spec_cons: tot_cons/(Avg_cargo*1000)-[kg diesel/kg cargo]
	Spec_cons_norm: Spec_cons/0.327-[kg diesel/kg cargo] [a]

[a] normalising factor for in-/out-flows: original dataset expressed per 0.327kg diesel (this value should change depending on the original dataset: 0.233kg diesel/km for 16t truck; 0.327kg diesel/km for 28t truck; 0.395kg diesel/km for 40t truck).

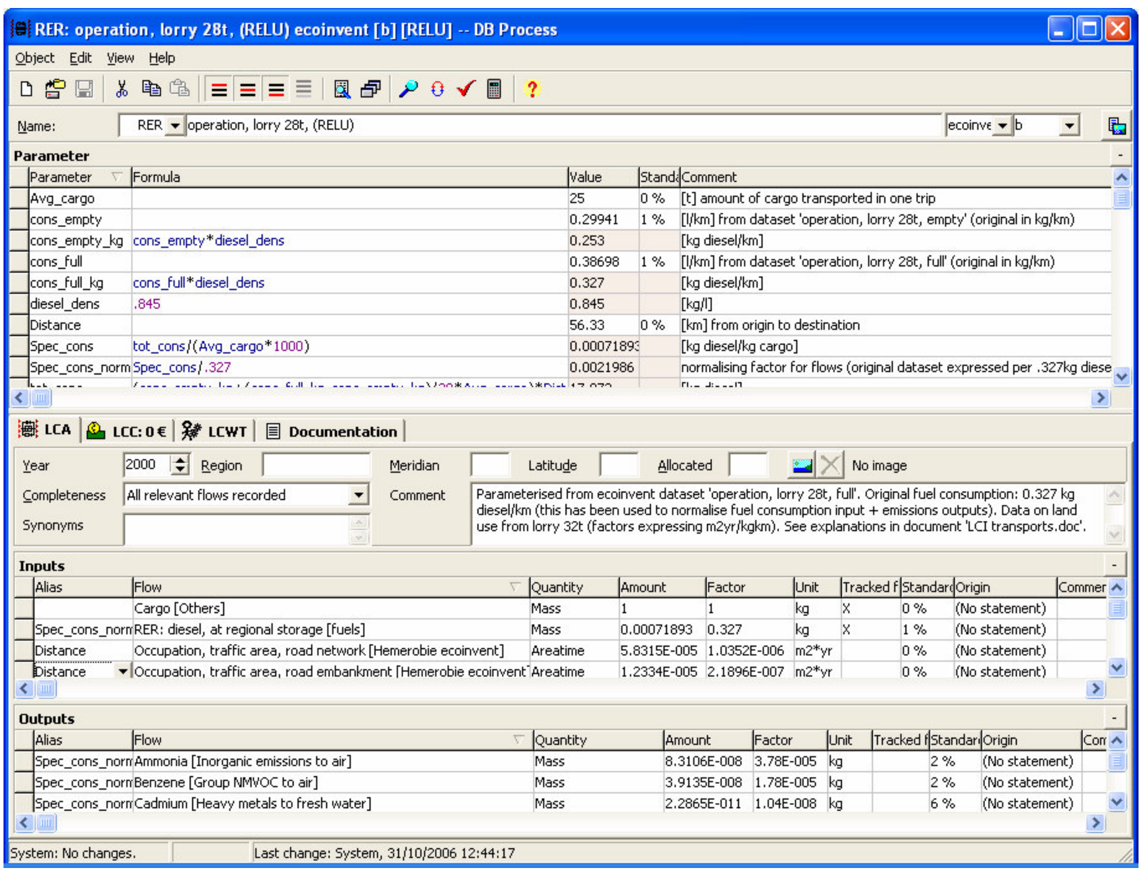

Figure 4-1: Example of fully parameterised truck dataset.

4.3.2 Operation of refrigerated trucks

For refrigerated trucks, the option to include the time spent in loading/unloading the truck and fuel use to keep the truck refrigerated during this process has been included (see Figure 4-2). However, it should be noted that in practice no big differences in total energy use have been found due to the inclusion of this loading/unloading fuel use.

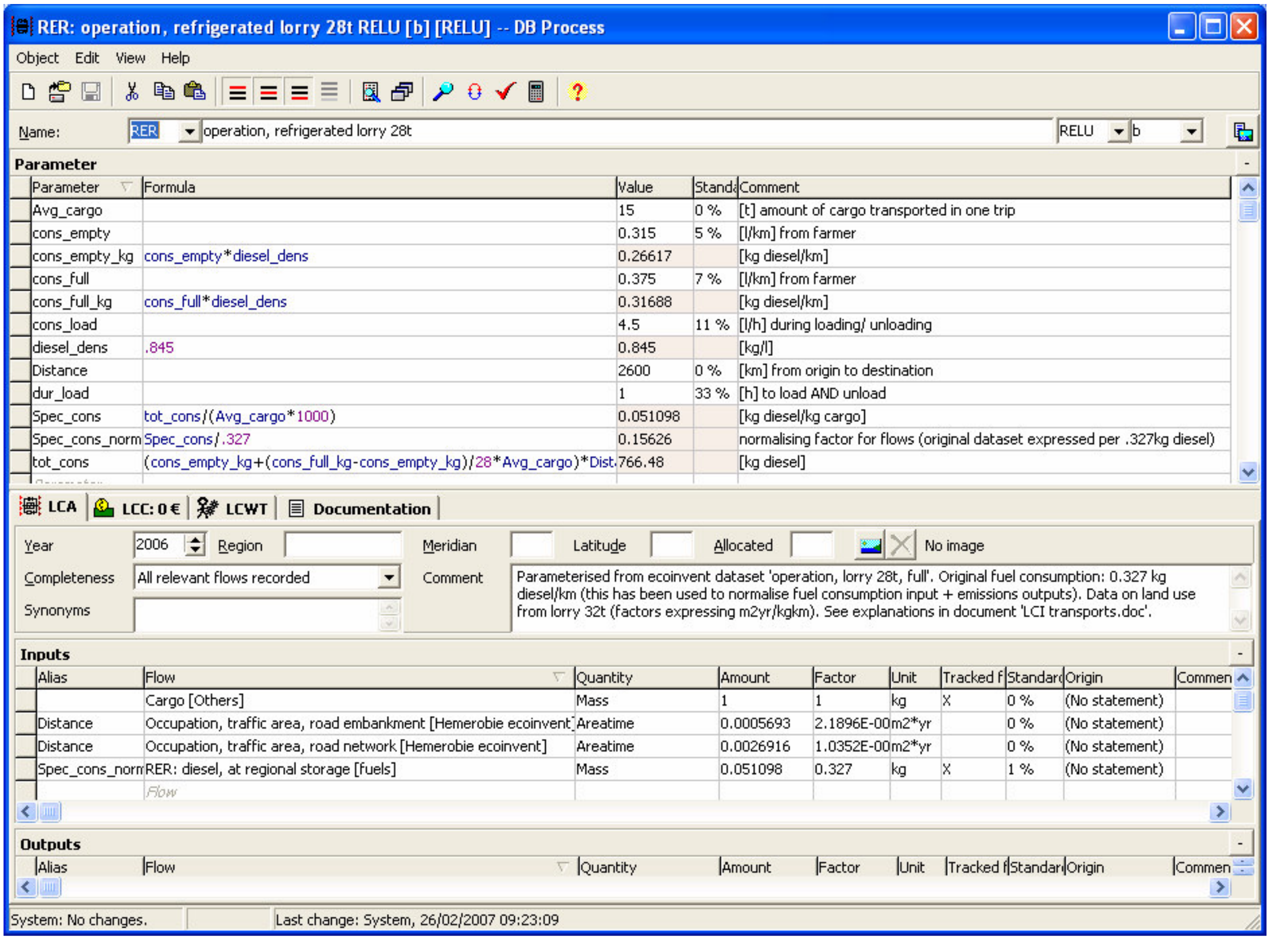

Figure 4-2: Example of fully parameterised refrigerated truck dataset.

4.3.3 Operation of mini-vans for fresh produce in Uganda

A special case of transport vehicle has been found in Uganda: mini-vans adapted for people transport ("taxis") are often used to transport fresh produce to the local markets or even to exporters. This has been adapted from he ecoinvent process 'operation, van < 3,5t'. Fuel consumption has been assumed to remain constant regardless of passenger occupation, and original dataset has been directly transformed to the delivery of 8pkm (instead of 1 vkm): normalising factor (cons_pkm) for original dataset (expressed per vkm; new in pkm assuming 8 passengers). The transportation of fresh produce also requires a transformation factor from kg produce to 'passenger km', e.g. two 40kg sacks of French beans have been considered to displace one passenger.

4.4 Operation of aircrafts (plane transportation)

In the case of cargo plane transportation, the recently available European Reference Life Cycle Database (ELCD) has been used instead of ecoinvent. The main reason for this is that ecoinvent datasets are mostly representative of intra-European flights, whereas the ELCD provides datasets for intercontinental flights. The ELCD datasets are already parameterised in GaBi 4.2, and therefore no further transformations have been deemed necessary; only the transportation distance has been adapted for this study (see section 5.1.1).

4.5 Electricity generation in Africa

As mentioned previously, the ecoinvent database does not provide datasets for power generation in African countries. A detailed assessment of environmental impacts of power generation in Africa was considered outside the scope of this project; however, the electricity generation mixes are so different in Africa compared to Europe that European datasets cannot be used straight away. Finally, impacts from individual power generation technologies have been used from ecoinvent, combined with the power generation mixes shown in Table 4-2.

Table 4-2: Electricity generation sources in Uganda and Kenya.

	Uganda [a]	***Kenya*** [b]
Hydroelectricity	95%	51%
Oil	5%	24%
Biomass		6%
Geothermal		19%

[a]: Actually, values of up to 99% hydro power in Uganda have been found. However, there are currently big investments in providing new thermal (normally coal-based) power plants in order to provide emergency power in case of drought-related power shortages.
[b] International Energy Agency (2007).

5 LIFE CYCLE INVENTORY (LCI) MODELLING FOR RETAIL TO PLATE OPERATIONS

IVAN MUÑOZ, LLORENÇ MILÀ I CANALS

This chapter focuses on the inventory analysis of the following life cycle stages:

- Distribution: this stage includes the transport of the packed products from the farm to the Regional Distribution Centre (RDC), and storage in the latter.
- Transport to the retailer and retailing operations.
- Transport by the consumer, home storage, and consumption.
- Management of solid waste generated in the retail and home stages.

Figure 5-1 gives an overview of the processes included in this report.

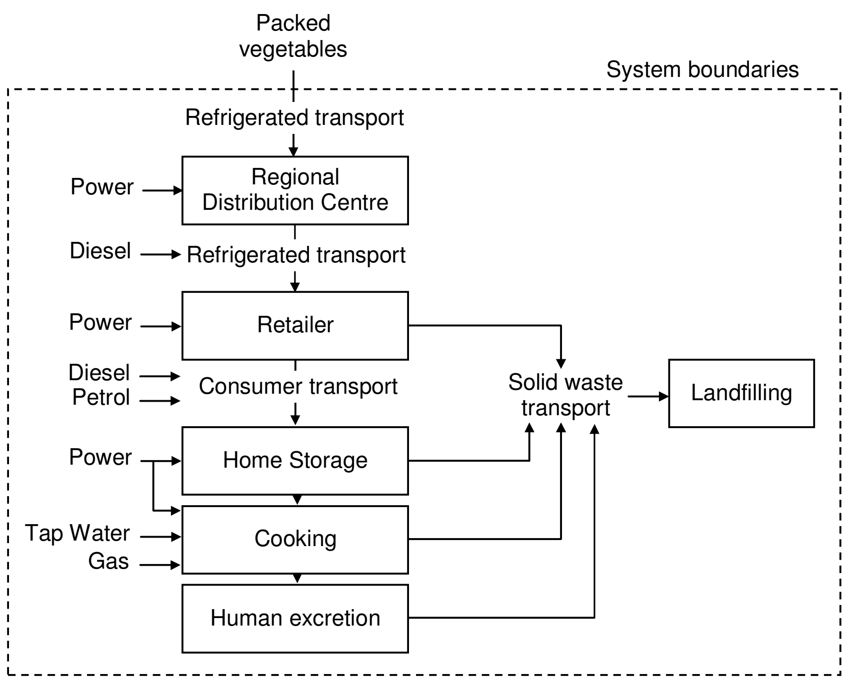

Figure 5-1: Overview of the processes modelled in the retail to grave stages of vegetables.

5.1 Distribution phase

5.1.1 Transport to RDC

After post-harvest operations and/or industrial processing, packed vegetables are transported to RDCs. This transport service is carried out by means of refrigerated trucks. Modelling of this type of transport in the RELU project is described in chapter 4. Concerning the transport distances, these have been determined specifically depending on the product and country of origin. Table 5-1 shows the distances considered.

Table 5-1: Distances for transport to RDC (km).

Product / origin	Leafy salads	Broccoli	Peas (frozen)	Beans
Spain (truck)	2600	2600	-	-
UK (truck)	200	200	200	200
Uganda (taxi/truck + plane + truck)	70 + 6,600 + 200	-	-	70 + 6,600 + 200
Kenya (plane + truck)	70 + 6,600 + 200	-	-	70 + 6,600 + 200

5.1.2 Storage in RDC

In the RDC, vegetables are stored prior to their final transport to the retail outlet. The main environmental issue of this operation is the energy consumed for cold or frozen storage. The latter has been allocated to the vegetables on the basis of volume occupied and storage time. The data on the specific energy consumption is from the Danish LCA Food Database (Nielsen, 2003), according to which 0.00059 kWh/L/day and 0.00063 kWh/L/day are consumed in wholesale facilities for cold and frozen storage, respectively. It is assumed that this operation in RDCs is similar to that in wholesalers, in terms of storage temperature, applied cooling technology, etc. Storage time has been set as 5 days for cooled vegetables and 30 days for frozen vegetables (Ritchie, 2005), while volume occupied is product-specific and has been determined from the packed product density (Table 5-2).

Table 5-2: Specific volume of packed vegetables

Product	Broccoli [a]	Beans [a]	Potatoes [b]	Lettuce [c]	Peas (frozen) [c]	Chicory [c]	Onions [c]
Specific volume (L/kg)	5.0	2.4	1.3	7.1	1.7	3.9	1.6
kg/m^3	200.0	416.7	769.2	140.8	588.2	256.4	625

[a] Ritchie (2005).
[b] www.simetric.co.uk/
[c] Own measurement with retailer samples.

The life cycle inventory for electricity production for Great Britain at medium voltage is taken from the Ecoinvent database, as supplied in the GaBi software.

With regard to food losses during storage, all the literature reviewed considers no losses at all in the RDC, or they are not estimated. In the present study no losses are taken into account either.

In the case of Ugandan and Kenyan produce, the storage process before plane transportation has been considered equal to the RDC. The only differences have been a storage time of 1 day and obviously the power mixes used.

5.1.3 Transport to retailer

This transport service is also carried out by means of refrigerated trucks, modelled as described in chapter 4. The transport distance is assumed for all vegetables as 50 km.

5.1.4 Storage and display in retail outlet

Retail includes two aspects in our model: energy use for storage/display of the product, and food losses.

Depending on the product type, it can be displayed in the retail outlet at ambient temperature, chilled, or frozen. The specific energy use of these options is shown in Table 5-3, and has been obtained from Danish and Swedish literature sources. The storage time is set as 2 days for products at ambient temperature or chilled, and 15 days for frozen products (Ritchie, 2005). For cooled products, the energy use depends on the specific product volume, which has already been displayed for the different vegetables in Table 5-2.

Table 5-3: Specific energy use for storage at retailer.

Storage	Energy Use	Source and comments
Ambient temperature	0.027 MJ/kg/day	Carlsson-Kanyama (1998). 44% electricity and 56% heating, assumed as natural gas in our study.
Cooled	0.06 MJ/L/day	Weidema et al. (1996). Product displayed in cold racks or cold counters. Assumed as 100% electricity.
Frozen	0.18 MJ/kg/day	Naturvårdsverket (1997). Electricity consumption for frozen food in retailer.

The life cycle inventory for electricity production for Great Britain at medium voltage is taken from the Ecoivent database, as supplied in the GaBi software.

Food losses at the retail stage have been considered to be 2% of the product sold, that is, 20 g product are lost per kg product sold (NFA, 1985, in Carlsson-Kanyama and Faist, 2000). The value of 2% has been considered both for fresh and frozen produce.

5.1.5 Land use in storage and retail stages

Carlsson-Kanyama (1998) offers values for storage density (kg/m^2) in retail stores, where she suggests only 5-10% of the space is used to actually display produce (the rest is used for alleys, tills, logistics area, car parks, etc.). Such values have been combined with the storage times from Ritchie (2005) to produce the following land occupation figures for retailer and RDC (Table 5-4):

Table 5-4: Land use associated to RDC and retailer for cool and frozen products.

	Storage density [kg/m^2]	Space used for produce [%]	Storage time [days]	Land occupation [m^{2}year]	Land type (from ecoinvent)
Cool produce, RDC	100	10	5	1.37E^{-3}	Industrial area, built up
Frozen produce, RDC	15	10	30	5.48E^{-2}	Industrial area, built up
Cool produce, retailer	100	5	2	1.10E^{-3}	Urban area, discontinuously built
Frozen produce, retailer	15	5	15	5.48E^{-2}	Urban area, discontinuously built

5.2 Consumer phase

5.2.1 Transport home

Vegetables are transported home by the consumer along with other products in the typical weekly shopping. This transport step has been modelled based on the work by Pretty et al. (2005). According to these authors, the average distance from the shop to home is 6.4 km, and the means of transport used is as follows:

- 58% of the trips are made by car,
- 30% by walking,
- 8% by bus,
- 3% by cycle

Only the trips by car and bus have been taken into account from an environmental impact perspective. It is assumed that car and bus trips are solely for shopping. An average shopping basket weight of 28kg (Foster *et al*. 2006) and an occupation of 30 people in the bus are considered.

The distances travelled per kg food are: 0.185 km by car, and 0.00085 km by bus. The model includes fuel production and combustion for each one of these transport modes. In the case of cars, the data used considers an average European passenger car, with 19% of the fuel consumed (in weight units) as diesel, and 81% as petrol. Concerning the bus, the Ecoinvent database does not include specific data for this type of transport. The fuel consumed by an urban bus is estimated at around 0.4 L/km (Öko-Institut, 2000), so in order to model the bus, a truck with similar fuel consumption has been used, namely a 40 ton truck, which is the biggest one available in the database.

5.2.2 Home storage

Preparation and storage of food within homes often accounts for important shares of the overall environmental impacts in the life cycle of food (Sonesson *et al*. 2003). Statistical data from the UK is available (Fawcett *et al*. 2005), but difficult to use to allocate energy use to specific products. The models suggested by Sonesson *et al*. (2003) are used as a first approximation to energy use for storage and preparation of food at home; these models only consider electric appliances because gas stoves or ovens are not common in Sweden, whereas they represent 54% and 41% of the appliances in the UK respectively (Fawcett *et al*. 2005). It has been considered that specific electricity consumption of Swedish appliances is representative of British appliances. Data for gas stoves and ovens has been estimated from the Swedish models and data from Fawcett *et al*. (2005). No land use has been associated to the home stage, as the main function of houses is to shelter humans.

Sonesson et al. (2003) offer models for three types of cold storage: chest freezers; upright freezers and refrigerators. The summary of their models is presented in the following equations, and the data used in the parameterisation of the models is presented in Table 5-5 for the different products considered in the study. The general characteristics considered for the cold storage appliances are presented in Table 5-6.

The energy use of freezer storage is derived from equations 1 to 3:

$$E_{chest\_freezer} = 4.36 \cdot V_{cabinet} \cdot \frac{D_{stored}}{365} \cdot \frac{V_{product}}{V_{used}} \quad [1]$$

$$E_{upright\_freezer} = 195.59 \cdot V_{cabinet}^{(-0.6429)} \cdot \frac{D_{stored}}{365} \cdot \frac{V_{product}}{V_{used}} \quad [2]$$

Where:
$E_{chest\_freezer}$ is the total energy use for chest freezer storage (MJ),
$E_{upright\_freezer}$ is the energy use for upright freezer storage (MJ),
$V_{cabinet}$ is the volume of the freezer cabinet (L),
D_{stored} is the time of storage in the freezer (days),
$V_{product}$ is the volume occupied by the product in the freezer cabinet (L), and
V_{used} is the volume of the freezer cabinet actually occupied by food (L).

If the product is not originally frozen when introduced into the freezer, a chilling component must be added to the freezer models:

$$E_{chill\_freezer} = wc_{product} \cdot m_{product} \cdot 3.34 \cdot 10^{-4} + Cp_{product} \cdot m_{product} \cdot \Delta T \quad [3]$$

Where:
$E_{chill\_freezer}$ is the total energy use required to freeze the product (MJ),
$wc_{product}$ is the water content of the product, in fresh weight (g/g),
$m_{product}$ is the mass of product to be frozen (g),
$Cp_{product}$ is the heat capacity of the product (MJ/g/ ºC)
ΔT is the temperature difference when freezing the product (38 ºC, decrease from 20 ºC to -18 ºC).

The energy use of fridge storage is derived from equation 4:

$$E_{refrigerator} = 371.59 \cdot V_{cabinet}^{(-0.8982)} \cdot \frac{D_{stored}}{365} \cdot \frac{V_{product}}{V_{used}} \quad [4]$$

Where:
$E_{refrigerator}$ is the total energy use for fridge storage (MJ),
$V_{cabinet}$ is the volume of the freezer cabinet (L),
D_{stored} is the time of storage in the freezer (days),
$V_{product}$ is the volume occupied by the product in the freezer cabinet (L), and
V_{used} is the volume of the freezer cabinet actually occupied by food (L).

Table 5-5: Parameters used in the cold storage models for the studied products

Parameter	Potatoes	Lettuce	Chicory	Broccoli	Frozen broccoli	Onions	Peas (frozen)	Green beans
% Fridge [a]	10	100	100	70	0	10	0	80
% Freezer [b]	0	0	0	20	100	0	100	20
% No cool [c]	90	0	0	10	0	90	0	0
Days in fridge	10	5	5	5	-	10	-	5
Days in freezer	-	-	-	15	15	-	15	15
Volume product [d]	1.3	7.1	3.9	5.0	5.0	1.6	1.7	2.4
Cp, heat capacity [e]	$3.4 \cdot 10^{-6}$	$4.1 \cdot 10^{-6}$	$4.0 \cdot 10^{-6}$	$3.8 \cdot 10^{-6}$	$3.8 \cdot 10^{-6}$	$3.9 \cdot 10^{-6}$	$3.4 \cdot 10^{-6}$	$3.9 \cdot 10^{-6}$
wc, water content [f]	0.79	0.951	0.943	0.882	0.882	0.89	0.746	0.907
Initial temperature	20ºC	20ºC	20ºC	20ºC	-4ºC	20ºC	-4ºC	20ºC

[a] % of product stored in fridge.
[b] % of product stored in freezer.
[c] % of product not cold-stored.
[d] Useful volume occupied by the product within the fridge/freezer cabinet [litres/kg]. From table 2.
[e] from (Sonesson et al. 2003) Table 27.
[f] Food Standards Agency (2002).

Table 5-6: Characteristics considered for the cold storage appliances

Parameter	Chest freezer	Upright freezer	Refrigerator
% freezers [a]	16%	84%	-
Average cabinet volume [litres] [b]	270	202	272
Used volume [litres] [c]	202	151	204

[a] From Fawcett et al. (2005); fridge-freezers have been considered as upright freezers; % based on total share of appliances, disregarding the fact that many households actually have more than one.
[b] From (Sonesson et al. 2003), Table 26.
[c] 75% has been considered.

5.2.3 Cooking

For cooking, four of the methods modelled by Sonesson et al. (2003) are used: boiling in water on hotplates; frying in a frying pan; roasting / baking in the oven; and microwaving. Different proportions of these methods are used for the different products (Table 5-7). As only data for electric appliances is offered in this reference, the direct energy consumed by gas ovens and hobs has been estimated using the following efficiency data from Fawcett et al. (2005):

- The energy use ratio of gas hobs/electric hobs is 1.51. This means that a gas hob uses 51% more energy than an electric hob in order to heat the same product.
- The energy use ratio of gas ovens/electric ovens is 1.39. This means that a gas oven uses 39% more energy than an electric oven in order to heat the same product.

In the case of boiling, the energy demand varies with the amount of water used for boiling and whether a lid is used or not. It has been assumed that more than 900g water are always used, and that no lid is in place.

Energy for frying depends on the temperature range (low; medium; high), as well as cooking time and surface of pan used. Medium temperature range has been used for the calculations, and the same size of pan has been considered for all products (95 cm^2); it should be noted that for big functional units this should be changed, or the frying time increased accordingly to allow for several batches. In general, electricity consumption also varies depending on the type of electric hob (cast iron / ceramic). As no data for share of households with cast iron/ ceramic hobs have been found, it has been assumed that cast iron hobs are in place.

The equations suggested by Sonesson et al. (2003) for these methods of cooking are presented below. For a detailed description of the models and their validity please see Sonesson et al. (2003). Table 5-7 provides the parameters used for the different products.

The energy use for boiling is derived from equation 5:

$$E_{boiling} = e_{hu,b} \cdot m_w + e_{mt,b} \cdot m_w \cdot t_b + c_p \cdot m_p \cdot \Delta T \qquad [5]$$

Where:
$E_{boiling}$ is the total energy used for boiling the product (MJ).
$e_{hu,b}$ is the energy for heating one gram of water to boiling point, including heating the hotplate and saucepan. Considered $5.8 \cdot 10^{-4}$ MJ g^{-1} from Sonesson et al. (2003), p.10 considering cast iron hob and no lid.
m_w is the amount of water used in boiling (g).
$e_{mt,b}$ is the energy for maintaining the boiling temperature of one gram of water for one minute. Considered $2.1 \cdot 10^{-5}$ MJ g^{-1} $minute^{-1}$ from Sonesson et al. (2003), p.10, considering cast iron hob and no lid.
t_b is the boiling time (minutes).
c_p is the capacity of the food (MJ g^{-1} $ºC^{-1}$). See Table 4.
m_p is the amount of product cooked (g).
ΔT is the mean temperature elevation in the product (ºC). Considered 78ºC for all products (increase from 20ºC to 98ºC).

The energy use for frying is derived from equation 6:

$$E_{frying} = m_{fp} \cdot \rho + e_{hu,f} \cdot A_{fp} + e_{mt,f} \cdot A_{fp} \cdot t_f \qquad [6]$$

Where:
E_{frying} is the total energy used for frying the product (MJ).
m_f is the mass of frying pan (g). Considered as 2382g.
ρ is the heat capacity of the pan. An iron pan is considered, with a ρ of $4.5 \cdot 10^{-7}$ MJ g^{-1} $ºC^{-1}$ (Sonesson et al. 2003).
$e_{hu,f}$ is the energy for heating the frying pan. Considered as $5.28 \cdot 10^{-3}$ MJ cm^{-2} (Sonesson et al. 2003, p.24), considering a cast iron hob at medium temperature.
A_{fp} is the area of the frying pan: 95 cm^2 are considered.
$e_{mt,f}$ is the energy for maintaining the temperature during frying for one minute. The value considered is $4.75 \cdot 10^{-4}$ MJ cm^{-2} $minute^{-1}$ (Sonesson et al. 2003, p.24), considering a cast iron hob at medium temperature.
t_f is the frying time (minutes).

The energy use for roasting is derived from equation 7:

$$E_{roasting} = e_{hu,r} \cdot V_o + e_{mt,r} \cdot V_o \cdot t_r + c_p \cdot m_p \cdot \Delta T + e_{ew} \cdot m_{wevap} + e_{tp} \cdot m_{frozen} \qquad [7]$$

Where:

$E_{roasting}$ is the total energy used for roasting the product (MJ).
$e_{hu,r}$ is the energy for heating one litre of oven volume. Considered $2.0 \cdot 10^{-4}$ MJ litre^{-1} ºC^{-1} (Sonesson et al. 2003, p.27).
V_o is the volume of the oven (L).
$e_{mt,r}$ is the energy for maintaining a certain oven temperature in one litre for one minute. Considered as $4.3 \cdot 10^{-6}$ MJ litre^{-1} ºC^{-1} minute^{-1} (Sonesson et al. 2003, p.27).
t_r is the roasting time (minutes).
c_p is the heat capacity of the food (MJ kg^{-1} ºC^{-1}). See Table 5-5.
m_p is the amount of product cooked (g).
ΔT is the mean temperature elevation in the product (ºC). Considered 180ºC for all products (from 20ºC to 200ºC).
e_{ew} is the energy for evaporating water. $2.26 \cdot 10^{-3}$ MJ g water evaporated^{-1} (Sonesson et al. 2003, p.28).
m_{wevap} is mass of water evaporated from the product (see Table 5-5)
E_{tp} is the melting energy for frozen food ($3.34 \cdot 10^{-4}$ MJ/g)
M_{frozen} is the mass of frozen food (g).

The energy use for microwaving is derived from equation 8:

$$E_{microw} = \frac{m_p \cdot \Delta T \cdot c_p + (m_{wevap} \cdot e_{ew})}{0.95 \cdot 0.86 \cdot 0.73 \cdot (0.57 + 3.8 \cdot 10^{-4} \cdot m_p)} \quad [8]$$

Where:
E_{microw} is the total energy used for microwaving the product (MJ).
m_p is the amount of product cooked (g).
ΔT is the mean temperature elevation in the product (ºC). Considered 78ºC for all products (increase from 20ºC to 98ºC).
c_p is the heat capacity of the food (MJ kg^{-1} ºC^{-1}). See Table 5-5.
m_{wevap} is mass of water evaporated from the product. This parameter is estimated for vegetables as 5% of m_p, according to experimental data in Sonesson et al. 2003).
e_{ew} is the energy for evaporating water ($2.26 \cdot 10^{-3}$ MJ/g).

Table 5-7: Parameters used in the cooking models

Parameter	**Potatoes**	**Lettuce**	**Chicory**	**Broccoli**	**Onions**	**Peas (frozen)**	**Beans**
% boiled	40%	0%	0%	80%	10%	70%	60%
% fried	20%	0%	0%	10%	60%	0%	10%
% roasted	30%	0%	10%	10%	30%	0%	0%
% microwaved	20%	0%	0%	0%	0%	30%	30%
Boiling time (minutes)	20	-	-	10	10	10	10
Boiling water (L/kg product)	1	-	-	5	5	5	5
Frying time (minutes)	20	-	-	15	15	-	15
Roasting time (minutes)	30	-	15	15	15	-	-
Microwave time (minutes)	15	-	-	-	-	5	5

Electricity consumption has been modelled with the Ecoinvent low voltage UK mix, while for natural gas consumption, there is not an Ecoinvent dataset representative of kitchen stove burning. The dataset used corresponds to burning natural gas in the smallest boiler included in the database (100 kW), taking into account natural gas extraction and transport as well.

5.2.4 Food losses

The amount of food not eaten in households is not very well known, as it is highly variable depending on the food product. Carlssson-Kanyama *et al*. (2001) suggest that 16% of potatoes bought are lost, while Carlsson-Kanyama and Faist (2000) give figures of 18% for cabbage, and 28% for carrots. For food in general, research commissioned by WRAP (2007) suggests that 1/3 of food bought is never eaten. In the RELU project we have considered for all fresh vegetables that 20% of the food input to the household leaves it as solid waste. Intuitively this value should be smaller for frozen vegetables, and although we have not found any published reference we have considered a 5% loss for frozen vegetables.

5.2.5 Nutrient losses to boiling water

During cooking, nutrients may be lost due to degradation, or to leaching, for example. In our case studies we have taken into account only the loss of nutrients when food is boiled. No losses are assumed to occur as a result of frying, baking or microwaving. Vitamins will clearly be lost in all these cooking methods, although these nutrients are not taken into account in food composition.

The loss of nutrients has been quantified by comparing the composition of vegetables before and after boiling (Table 5-8). This table excludes all the vegetables for which boiling is not considered, and also green beans, which show no losses according to Food Standards Agency (2002). As it can be seen, the losses are quite remarkable; as an example, broccoli losses 30% of its initial protein and carbohydrate content.

Table 5-8: Estimation of nutrient losses due to boiling of some vegetables.

Parameter	Potatoes			Broccoli			Peas (frozen)		
	Raw	**Boiled**	**Boiling loss**	**Raw**	**Boiled**	**Boiling loss**	**Raw**	**Boiled**	**Boiling loss**
Water (g/100 g)	0.79	80.3		88.2	91.1		74.6	75.6	
Main organic constituents:									
Protein (g/100 g)	2.1	1.8	0.3	4.4	3.1	1.3	6.9	6.7	0.2
Fat (g/100 g)	0.2	0.1	0.1	0.9	0.8	0.1	1.5	1.6	-0.1
Carbohydrate (g/100 g)	17.2	17	0.2	1.8	1.1	0.7	11.3	10	1.3
Fibre (g/100 g)	1.3	1.2	0.1	2.6	2.3	0.3	4.7	4.5	0.2
Inorganic constituents:									
P (g/100 g)	0.037	0.031	0.006	0.087	0.057	0.03	0.13	0.13	0
Na (g/100 g)	0.007	0.007	0	0.008	0.008	0	0.001	0	0.001
K (g/100 g)	0.36	0.28	0.08	0.37	0.17	0.2	0.33	0.23	0.1
Ca (g/100 g)	0.005	0.005	0	0.056	0.056	0	0.021	0.019	0.002
Mg (g/100 g)	0.017	0.014	0.003	0.022	0.022	0	0.034	0.029	0.005
Cl (g/100 g)	0.066	0.045	0.021	0.1	0.1	0	0.039	0.008	0.031

Source: Food Standards Agency, 2002.

These amounts of nutrients are discharged to the sewer along with the boiling water. The overall volume of wastewater is determined from the amount of boiling water used (Table 5-7), and the environmental burdens of treating these wastewaters are determined with the wastewater treatment model described in Muñoz et al. (2007).

5.2.6 Human excretion

The human body can be seen as a biochemical reactor, which processes ingested food to obtain energy and gives rise to different pollutants released to air and water, which should be included within the system boundaries of a complete LCA, in a similar way as it is done when food waste is landfilled or composted. This is particularly relevant in attributional food LCA in order to identify the life cycle hot spots.

However, up to date, little attention has been paid to this life cycle stage of food products, which is usually omitted. A complete and systematic procedure for including this process in LCA studies is lacking; for this reason, a specific model for human excretion has been developed within the framework of the RELU project, and a detailed description can be found in Muñoz et al. (2007).

This model calculates the following environmental burdens:

- Emissions to air from digestion and respiration: carbon dioxide, water, and methane.
- Emissions to wastewater from digestion and renal excretion: nitrogen, phosphorus, COD, inorganic elements, etc. These are treated in a specific module for wastewater treatment, including the energy, chemicals, infrastructure, and final emissions to the environment of wastewater and sludge treatment.
- Auxiliary materials and energy related to toilet use: tap water, toilet paper, soap, among others.

The input to this excretion model is the composition of the food to be assessed. The following table shows the composition considered for each vegetable (Food Standards Agency, 2002). An inventory for the whole excretion-wastewater treatment system has been obtained for each vegetable.

Table 5-9: Composition of food as introduced to the human excretion model

Parameter	Potatoes				Lettuce	Chicory	Broccoli	Onions		Peas (frozen)		Green Beans	
Cooking method	**Raw**	**Baked**	**Fried**	**Boiled**	**Raw**	**Raw**	**Boiled**	**Fried**	**Raw**	**Raw**	**Boiled**	**Raw**	**Boiled**
Water (g/100 g)	79.0	78.9	56.5	80.3	95.1	94.3	91.1	65.7	89.0	74.6	75.6	90.7	90.0
Main organic constituents:													
Protein (g/100 g)	2.1	2.2	3.9	1.8	0.8	0.5	3.1	2.3	1.2	6.9	6.7	1.7	1.7
Fat (g/100 g)	0.2	0.1	6.7	0.1	0.5	0.6	0.8	11.2	0.2	1.5	1.6	0.5	0.1
Carbohydrate (g/100 g)	17.2	18	30.1	17	1.7	2.8	1.1	14.1	7.9	11.3	10	3.2	4.7
Fibre (g/100 g)	1.3	1.4	2.2	1.2	0.9	0.9	2.3	3.1	1.4	4.7	4.5	2.2	4.1
Inorganic constituents:													
P (g/100 g)	0.037	0.04	0.062	0.031	0.028	0.027	0.057	0.044	0.03	0.13	0.13	0.038	0.038
Na (g/100 g)	0.007	0.007	0.012	0.007	0.0003	0.001	0.008	0.004	0.003	0.001	0	Tr	0
K (g/100 g)	0.36	0.36	0.66	0.28	0.22	0.17	0.17	0.37	0.16	0.33	0.23	0.23	0.16
Ca (g/100 g)	0.005	0.007	0.011	0.005	0.028	0.021	0.056	0.047	0.025	0.021	0.019	0.036	0.056
Mg (g/100 g)	0.017	0.018	0.031	0.014	0.006	0.0006	0.022	0.008	0.004	0.034	0.029	0.017	0.017
Cl (g/100 g)	0.066	0.072	0.12	0.045	0.043	0.025	0.1	0.053	0.025	0.039	0.008	0.009	0.021

Source: Food Standards Agency (2002).

5.3 WASTE MANAGEMENT

The main disposal route for food waste in the UK is sanitary landfilling, rather than incineration. DEFRA figures on solid waste management show that in the 2005-2006 period, 64% of solid waste arisings in the UK were managed by landfilling, while only 8% was incinerated (DEFRA, 2007). In addition, we have made an estimation of the percentage of food waste treated by composting in England: according to DEFRA (2007), England produces 25.5 million tones/year of solid waste, of which 17% is kitchen waste, i.e. 4.335 million tonnes/year, and 18% is garden waste, i.e. 4.59 million tones/year. The amount of waste composted in 2004, according to The Composting Association (2006), is 2.67 million tones, but from these, only around 68000 tones correspond to kitchen waste[4]. As a consequence, only 68000/4335000 = 1.6% of food waste is composted.

Due to the relatively low share of both composting and incineration of kitchen waste in the UK, we have considered sanitary landfill as the only disposal route for food waste in all RELU case studies.

5.3.1 Transport

Transport to the solid waste treatment facility has been modelled using Ecoinvent data. The average distance to sanitary landfill considered in Ecoinvent is 10 km, which is representative of the Swiss situation; we have used a distance of 50km in our model, with a 16 ton truck.

5.3.2 Sanitary landfill

Two types of waste must be taken into account in the model:

- Packaging waste: product packaging, namely polyethylene film and/or cardboard boxes
- Vegetable waste: losses through the distribution chain, as well as kitchen waste

Packaging waste landfilling has been modelled with an Ecoinvent dataset for sanitary landfilling of polyethylene, while for vegetables, no such dataset is available. For this reason, a specific inventory for sanitary landfilling of vegetables has been obtained with the Ecoinvent tool for landfills (Doka, 2003), using the elemental waste composition in Table 5-10, which is based on the composition of raw broccoli. This elemental composition has been in turn been obtained with the elemental composition of food constituents included in the human excretion model (Muñoz et al. 2007).

Table 5-10: Elemental composition considered for sanitary landfilling of vegetables

Water	O	H	C	S	N	P
0.9004	0.0369	0.0074	0.0472	0.0007	0.0065	0.0009

[4] 3000 tones of kitchen waste from kerbs + 2000 tones from commercial outlets + 128000 tones of garden and kitchen waste from kerbside collection; according to DEFRA (2007) 49% of organic household waste is kitchen waste, therefore the 128000 tones correspond to 62720 tones of kitchen waste.

6 ALLOCATION ISSUES FOR FOOD LCA

LLORENÇ MILÀ I CANALS

6.1 Land occupation and diffuse emissions

The studied systems present in many occasions allocation/system expansion requirements, as is common in biotic production systems (Milà i Canals *et al.*, in preparation). All studied crops are grown in a crop rotation, where operations performed during one crop often benefit others; some may actually be considered successional crops harvested in the same field during the same cropping year; and examples of intercropping (different crops using the same field at the same time) have been found in Uganda (maize and beans).

Several interventions could be allocated to one crop or another in a more or less sophisticated fashion as exemplified in Milà i Canals *et al.* (in preparation). Relevant interventions here include land occupation, nutrient inputs benefiting more than one crop, CO_2 emissions from degradation of soil organic carbon, and nutrient related emissions calculated on a per ha basis (NO_3^-; PO_4^{3-}). In these cases, the interventions have been equally allocated to all relevant crops on a per crop basis; i.e. if 2 crops are obtained per ha per year, then each one is attributed 0.5 ha*year of land occupation and half the CO_2; NO_3^-; and PO_4^{3-} emissions estimated per ha per year (see sections 3.4 and 3.5). When the farmer specifically states that e.g. base fertiliser is only applied before the first crop but it is intended to benefit both, the amount is also equally allocated between the 1st and 2nd crops. In terms of crop rotation, no operations have been identified that should be allocated to only one crop in the rotation.

6.2 Food waste

Food waste is a very important environmental issue of the food supply chain. Apart from the environmental impacts derived from e.g. production of un-eaten food and emissions from its degradation, it presents modelling challenges. Basically, the following cases have been identified and addressed in the following way in this project:

- Food waste affecting the reference flow to be assessed: e.g. if 20% of produce is wasted during storage, then production of 1.25kg of crop needs to be considered when assessing the delivery of 1kg to the consumer, as discussed and further exemplified by Milà i Canals *et al.* (2007d). The impacts derived from the degradation (e.g. in landfill) of such waste are also allocated to the studied system as explained in 5.2.4 and 5.3.
- Second grade produce or food wastage not affecting the reference flow. In some occasions a portion of food is wasted but re-used for other purposes (e.g. animal feed or in the food industry[5]). In these cases, it has been assumed that an equivalent product is displaced by the “waste” (by-product) flow, and a mass allocation has been used; this means that only the production of e.g. 1kg of raw produce has been considered necessary to produce 1 kg of finished (processed) produce. Considering an economic allocation in these cases could have a significant effect on the results as illustrated in Milà i Canals *et al.* (2007d) for fresh vs. process apples.

[5] E.g. one of the food processors preparing broccoli to be sold in florets rather than whole said that broccoli stems are bought by vegetable stock and soup manufacturers.

7 LIFE CYCLE IMPACT ASSESSMENT (LCIA) METHODOLOGY SPECIFIC TO THIS PROJECT

LLORENÇ MILÀ I CANALS, MIGUEL BRANDÃO

7.1 Spatial dependency

LCA was originally conceived as a site-independent environmental assessment, mainly due to data availability and the nature of the assessment (Milà i Canals 2003). Indeed, in industrial product development (the application for which LCA was conceived) mainly the technology type needs to be assessed with LCA, which advocates for a site-independent analysis: it is not practical or even possible to collect site-specific information for all sites included in the LCA (Finnveden and Nilsson 2005). Nevertheless, this may not hold true for some applications of LCA, and particularly for some sectors such as agriculture.

Site- (or spatial-) dependency has effects on the LCA goal and scoping, LCI and LCIA, and it may be considered from the point of view of application, sector, or impact category.

Wenzel (1998) discusses on the issue of site-dependency as related to the type of decision to be made based on the LCA results (application). According to him, three key variables determine the need for site-dependency: the nature and extent of the environmental consequences of the decision (including the occurrence of trade-offs between impact categories); the social and economic consequences; and the context of the decision. He concludes that the LCA applications needing to be site-dependent are mainly production technology assessment (Best Available Technologies), choice between alternative suppliers, and marketing. Oppositely, he suggests that LCA applied to societal activities planning and legislation, product development, and eco-labelling criteria setting would not need to be site-dependent. Ross and Evans (2002) maintain that excluding temporal and spatial site-dependent information to support decision making at the policy making level reduces the usefulness and credibility of LCA results. Finnveden and Nilsson (2005) mention regional planning as one application of LCA for which there is generally the required information for site-dependency, and where it is relevant for the LCIA to be site-dependent. Owens (2002) also suggests that the goal of the LCA and the sector where it is applied should be considered when deciding the specific impact indicators to assess the impacts on water resources.

Other authors have pointed out that, regardless of the application of the results, spatial-dependency is needed for some impact categories (particularly those having effects at regional or even local levels). Potting et al. (1998) derive country-dependent characterisation factors for acidification, and Huijbregts et al. (2000) do so for acidification and terrestrial eutrophication. Krewitt et al. (2001) derive characterisation factors for SO_2, NO_x, fine particles and NMVOC for impacts on several local and regional impact categories (human health, acidification, eutrophication and man-made environment). They conclude that including site-dependent data in the assessment results in a significant variation in the damage factors. Finnveden and Nilsson (2005) further determine that for some impacts (related to human health) a sub-country level of detail in site-dependency might be relevant to consider, whereas for impacts on ecosystems from SO_x and NO_x (acidification, eutrophication) a country level of detail may be enough. Owens (2002) discusses possible indicators to assess water quantity and quality in LCA, recognising the need for site-dependent information for many of them (although the author then suggests that this site-dependency should be considered using tools more suitable than LCA).

In the case of the agricultural sector, the environmental consequences related to agricultural systems depend on both the technology and the site where agricultural production takes place. Cowell and Clift (1998) suggest that site-dependent aspects might have a greater influence on the LCA results than activity-dependent aspects, which is confirmed in Milà i Canals (2003) and Milà i Canals et al. (2006). The LCAnet Food project also identified the role of geographical variations in the agricultural LCA results as a research priority, but this could not be satisfactorily addressed during the project (Olsson 1999). This influence may be derived from the inventory results (e.g.: on the substances emitted in different locations, which are affected by site characteristics such as soil and climate), or from the impact assessment results (e.g. through the effects on local impacts such as acidification, eutrophication, land use impacts, toxicity, etc.).

This project compares agricultural production in countries where the environmental conditions differ substantially. Therefore, the issue of spatial-dependency for regional impacts such as acidification, eutrophication and water use is very relevant. However, no data for characterisation factors in Uganda and Kenya have been found, and the main effects on such impact categories have only been checked in the case studies for the UK and Spain.

7.2 Land use impacts

The ecoinvent database reports detailed land use flows for all its datasets. This has been seized to answer one of the many research questions addressed in this project: what is the relative land occupation associated to the different life cycle stages of horticultural produce production? Data from ecoinvent have been combined with data compiled during the study related to the cropping stage and subsequent food storage, processing (e.g. packing) and retailing. In addition to the mere quantification of land occupation (measured in m^2year), land use impacts on soil quality have been calculated in the project.

Soil quality refers to the ability of soil to sustain life support functions (Milà i Canals 2003): biotic production; substance cycling and buffer capacity; climate regulation. The impacts of production systems on soil quality have not traditionally been included in LCA, and the recommendations of (Milà i Canals *et al.* 2007b; 2007c) have been followed in this study. Particularly, Milà i Canals *et al.* (2007c) argue that soil organic matter (SOM, often measured by soil organic carbon: SOC) can be used as an indicator for soil quality within LCA of agricultural systems.

This impact category refers mainly to the agricultural stage, as this will cause the main effects on soil quality. An increase in soil organic matter due to the soil management practices implies a benefit, whereas any decrease in SOM is accounted as damage from the system. The impact is measured in the unit ‘kg C·year’, referring to the amount of carbon temporarily present or absent from the soil due to the studied system (Milà i Canals *et al.* 2007c).

The calculation of these impacts requires measurements / estimates of the effects of the production system on SOC. The general formula used to calculate characterisation factors (CF) for land use flows is shown in Equation 7-1; see Figure 7-1 to see an illustration of the formula’s parameters.

Equation 7-1:

$$C\ deficit\left[kg\,C\cdot yr\cdot m^{-2}\cdot yr^{-1}\right]=\frac{\left(SOC_{pot}-SOC_{ini}\right)\times\left(t_{relax}-t_{ini}\right)+\frac{1}{2}\left(t_{relax}-t_{ini}\right)\times\left(SOC_{ini}-SOC_{fin}\right)}{\left(t_{fin}-t_{ini}\right)}$$

where SOC_{pot} is the potential level of SOC if land is left undisturbed; SOC_{ini} the SOC level at the start of the land use studied; SOC_{fin} is the SOC level at the end of the cultivation period; t_{ini} is the moment when the studied land use starts; at t_{fin} the land use finishes; at t_{relax}, is when soil quality has reverted to the level prior to land use; and $t_{relax,pot}$ is the time when the system reaches its potential quality. t_{relax} may be calculated from the relaxation rate R (see below). The equation assumes very simplified shapes of the evolution of soil quality, as suggested in Milà i Canals *et al.* (2007b). The first component of the numerator refers to the impacts due to the postponed relaxation of the system (vertical dashed area in Figure 7-1), whereas the second component is the “triangle” with horizontal dashes, referring to the impacts due to the change in quality during the occupation. The denominator serves to express the characterisation factors per $m^{-2}yr^{-1}$ (all the SOC values are expressed per m^2).

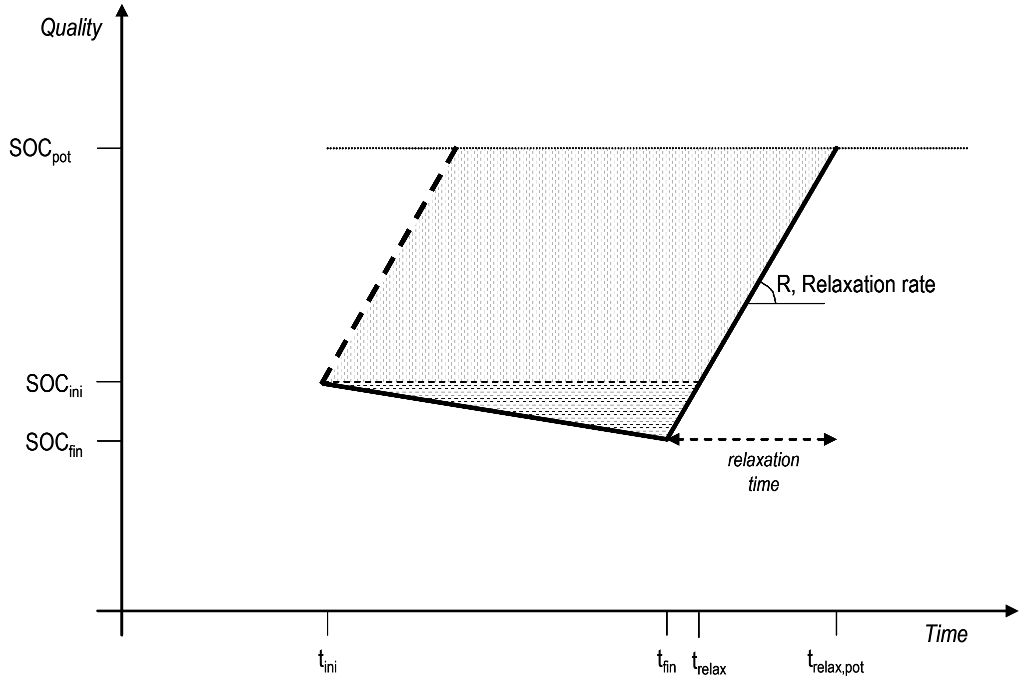

Figure 7-1: Calculation of impacts on soil quality measured by SOC (adapted from Milà i Canals *et al.* 2007b; 2007c).

The combination of the land use impacts from the cropping stage, where SOM measurements were available, with the rest of the land used through the rest of the life cycle (background land uses, from ecoinvent) requires the development of CF for such land use flows (both occupation and transformation). The following assumptions have been made to derive such CF:

- Occupation flows: no change in soil quality during the land occupation has been assumed (i.e. SOC_{ini}=SOC_{fin} and t_{fin}=t_{relax} in Figure 7-1). Only the first component of Equation 7-1 is thus needed to calculate CF for occupation. When expressed per m^2year this equals SOC_{pot}-SOC_{ini} if these parameters are expressed in kg C m^{-2}.
- Transformation flows: They are expressed as groups of two flows in ecoinvent: "transformation from land use x" and "transformation to land use y". To calculate the impacts on SOC the same reference (Potential state) has always been used (i.e. "transformation from land use x *to Potential*" and "transformation *from Potential* to land use y". Consequently, when such pairs of flows are combined, the impacts calculated for a specific dataset correspond to the real transformation (e.g. "*from x to y*"). For these flows, basically the second component of Equation 7-1 is required. For the sake of simplicity, a pulse transformation of 1 year duration is always assumed.
- The potential state (SOC_{pot}) for all background uses (both occupation and transformation) is 150 t C ha^{-1} (temperate warm forest, Bradley *et al.* 2005).

Table 7-1 provides the characterisation factors for the soil quality impact category for ecoinvent land occupation flows, and Table 7-2 does so for the ecoinvent land transformation flows.

Table 7-1: CF for occupation flows in ecoinvent (kgC yr m^{-2} yr^{-1})

Ecoinvent flow	MgC ha^{-1} SOC(fin)	MgC ha^{-1} SOC(Pot)	kgC yr m^{-2} yr^{-1} C deficit CF
Occupation, arable, non-irrigated	53	150	**9.70**
Occupation, construction site	0	150	**15.00**
Occupation, dump site	0	150	**15.00**
Occupation, dump site, benthos	excluded		
Occupation, forest, intensive	130	150	**2.00**
Occupation, forest, intensive, normal	130	150	**2.00**
Occupation, industrial area	2	150	**14.80**
Occupation, industrial area, benthos	excluded		
Occupation, industrial area, built up	0	150	**15.00**
Occupation, industrial area, vegetation	40	150	**11.00**
Occupation, mineral extraction site	0	150	**15.00**
Occupation, pasture and meadow, extensive	100	150	**5.00**
Occupation, pasture and meadow, intensive	100	150	**5.00**
Occupation, permanent crop, fruit, intensive	110	150	**4.00**
Occupation, shrub land, sclerophyllous	54	62.31	**0.83**
Occupation, traffic area, rail embankment	30	150	**12.00**
Occupation, traffic area, rail network	0	150	**15.00**
Occupation, traffic area, road embankment	30	150	**12.00**
Occupation, traffic area, road network	0	150	**15.00**
Occupation, urban, discontinuously built	4	150	**14.60**
Occupation, water bodies, artificial	excluded		
Occupation, water courses, artificial	excluded		

Table 7-2: CF for transformation flows in ecoinvent (kgC yr m^{-2}). Values not explained in footnotes are assumptions of this study.

Ecoinvent flow	years t_{relax}	MgC ha^{-1} SOC(fin)	MgC ha^{-1} SOC(ini)	kgC yr m^{-2} C deficit CF
Transformation, from arable	100	150	53 [a]	**-485**
Transformation, from arable, non-irrigated	100	150	53 [a]	**-485**
Transformation, from arable, non-irrigated, fallow	100	150	53 [a]	**-485**
Transformation, from dump site, inert material landfill	500	150	0	**-3750**
Transformation, from dump site, residual material landfill	500	150	0	**-3750**
Transformation, from dump site, sanitary landfill	500	150	0	**-3750**
Transformation, from dump site, slag compartment	800	150	0	**-6000**
Transformation, from forest	20	150	130 [b]	**-20**
Transformation, from forest, extensive	20	150	130 [b]	**-20**
Transformation, from industrial area	1000	150	2 [c]	**-7400**
Transformation, from industrial area, benthos	excluded	excluded	excluded	
Transformation, from industrial area, built up	1000	150	0	**-7500**
Transformation, from industrial area, vegetation	200	150	40 [d]	**-1100**
Transformation, from mineral extraction site	1000	150	0	**-7500**
Transformation, from pasture and meadow	50	150	100 [e]	**-125**
Transformation, from pasture and meadow, extensive	50	150	100 [e]	**-125**
Transformation, from pasture and meadow, intensive	50	150	100 [e]	**-125**
Transformation, from sea and ocean	excluded	excluded	excluded	
Transformation, from shrub land, sclerophyllous	50	62.31 [f]	54 [g]	**-21**
Transformation, from unknown	355 [h]	145 [h]	48 [h]	**-1721**
Transformation, to arable	100	53 [a]	150	**485**
Transformation, to arable, non-irrigated	100	53 [a]	150	**485**
Transformation, to arable, non-irrigated, fallow	100	53 [a]	150	**485**
Transformation, to dump site	500	0	150	**3750**
Transformation, to dump site, benthos	excluded	excluded	excluded	
Transformation, to dump site, inert material landfill	500	0	150	**3750**
Transformation, to dump site, residual material landfill	500	0	150	**3750**
Transformation, to dump site, sanitary landfill	500	0	150	**3750**
Transformation, to dump site, slag compartment	800	0	150	**6000**
Transformation, to forest	20	130 [b]	150	**20**
Transformation, to forest, intensive	20	130 [b]	150	**20**
Transformation, to forest, intensive, normal	20	130 [b]	150	**20**
Transformation, to heterogeneous, agricultural	71 [i]	81 [i]	150	**245**
Transformation, to industrial area	1000	2 [c]	150	**7400**
Transformation, to industrial area, benthos	excluded	excluded	excluded	
Transformation, to industrial area, built up	1000	0	150	**7500**
Transformation, to industrial area, vegetation	200	40 [d]	150	**1100**
Transformation, to mineral extraction site	1000	0	150	**7500**
Transformation, to pasture and meadow	50	100 [e]	150	**125**
Transformation, to pasture and meadow, extensive	50	100 [e]	150	**125**
Transformation, to pasture and meadow, intensive	50	100 [e]	150	**125**
Transformation, to permanent crop, fruit, intensive	50	110	150	**100**
Transformation, to sea and ocean	excluded	excluded	excluded	
Transformation, to shrub land, sclerophyllous	50	54 [g]	62.31 [f]	**21**
Transformation, to traffic area, rail embankment	500	0	150	**3750**
Transformation, to traffic area, rail network	500	0	150	**3750**
Transformation, to traffic area, road embankment	500	0	150	**3750**
Transformation, to traffic area, road network	700	0	150	**5250**

Transformation, to unknown	380 [h]	44 [h]	147 [h]	**1953**
Transformation, to urban, discontinuously built	1000	4	150	**7300**
Transformation, to water bodies, artificial	excluded	excluded	excluded	

[a] Arrouays *et al*. 2002
[b] Bradley *et al*. (2005) Table 6 woodland UK 0-30 cm
[c] Assuming 5% vegetated
[d] Bradley *et al*. (2005) Table 6 gardens UK 0-30 cm
[e] Bradley *et al*. (2005) Table 6 pasture UK 0-30 cm
[f] Forests in South Eastern Spain, data provided through personal communications with Dr Joan Romanyà May 2007
[g] Tropical woodland and savannah in Post *et al*. 1982
[h] Average values
[i] Average values of agricultural uses

7.3 Water use impacts

Water use derived impacts have seldom been considered in LCA, and their omission is currently considered one of the most important limitations of the method. This omission is particularly crucial in the agricultural sector, and when foodstuffs from different regions are being compared. For this project, quantification of the water resources used in different life cycle stages (particularly in cropping) has been used as a starting point. From the mere LCI quantification, methods used for virtual water (VW) calculations have been explored, and LCIA methods developed. This is further explained and illustrated in Milà i Canals *et al*. (in preparation a: b).

7.4 Pesticide use impacts

Several studies point at synthetic pesticides as the main cause of toxic effects on humans and ecosystems in horticultural production (Antón *et al*. 2003; Notarnicola *et al*. 2003; Milà i Canals *et al*. 2006; Hauschild *et al*. 2007; Margni *et al*. 2002). In parallel, pesticide rating systems have been developed to guide farmers' decisions towards active ingredients with potentially less harmful consequences (e.g. Kovach *et al*. 1992).
A gap in previous research is the assessment of simplified modelling approaches for pesticides in LCA. Existing approaches are either too simplistic (e.g. Audsley *et al*. 1997) and assume fixed rates of emission for pesticides, or demand large amounts of parameters on the pesticides used, the application techniques and the place of application (e.g. Klein 1995; Jarvis 1998; Geisler *et al*. 2004; Birkved and Hauschild 2006). The latter render very detailed and robust results, but are often impractical for LCA screening studies. Additionally, methods more commonly used by farmer advisors or farm extension services have not been checked for their adequacy in LCA studies. In particular, pesticide rating methods are a simple and valuable way to support decision making by farm advisors on a daily basis, but it is not known whether the results suggested by such methods are in line with e.g. what could be expected applying more sophisticated fate and exposure models.

7.4.1 The Environmental Impact Quotient

The Environmental Impact Quotient (EIQ) (Kovach *et al*. 1992) index has been chosen as a means of estimating hazards related to pesticide use in this project for three main reasons (Cross and Edwards-Jones, submitted):

1. It has the ability to report the hazard to a range of end-points separately (the end-points are farm-worker, consumer and environment);
2. it performed reasonably well in the comparative analysis of risk indices undertaken by Maud et al. (2001);
3. it has already been used in a number of other agricultural and horticultural contexts (Gallivan et al. 2001; Maud et al. 2001; Smith et al. 2002; Bues et al. 2004; Brimner et al. 2005) which serves to instil some confidence in its utility.

As described by Cross and Edwards-Jones (2006), the EIQ is a dimensionless value providing a rating for the inherent hazard of a pesticide to three non-target groups; farm worker, consumer and environment. A pesticide's EIQ rating is the product of its component parts $EIQ_{farmworker}$, $EIQ_{consumer}$ and $EIQ_{ecological}$. It is the mean of these three components that denotes the overall EIQ for a given pesticide (see Fig 1 in Cross and Edwards-Jones, 2006).

7.4.2 Use of the Environmental Impact Quotient in the LCA studies

EIQ values for the active ingredients used in the studied farms have been used as characterisation factors for a new impact category 'Pesticide Hazard'. EIQ values are thus multiplied by the doses of active ingredients (expressed per functional unit: kg of produce; hectare; etc.) to derive the expected pesticide hazard from different production systems (farms).

As this impact category refers only to the agricultural stage it is of no use to assess the relative toxicity impacts of this stage with the other stages in the life cycle (e.g. fertiliser production; distribution; etc.). It is however useful as an indicator of the relative toxicity of different production systems, as it includes the hazard to the farm worker, the consumer and the environment.

Milà i Canals *et al*. (in preparation c) explore the relationship of results obtained with the EIQ and other, sophisticated, LCA approaches.

REFERENCES

Althaus H-J, Blaser S, Classen M, Jungbluth N (2004) Life Cycle Inventories of Metals. Final report ecoinvent 2000 No. 10, Swiss Centre for Life Cycle Inventories, Duebendorf (Switzerland), for ecoinvent members only.

Antón, M.A., Castells, F., Montero, J.I., Muñoz, P., 2003. Most significant substances of LCA to Mediterranean Greenhouse Horticulture. In: Halberg, N. (Ed.), Life Cycle Assessment in the Agri-food sector. Proceedings from the 4th International Conference, October 6-8, 2003, Bygholm, Denmark. DIAS Report no. 61, Tjele, Denmark, pp. 3-7.

Arrouays D, Balesdent J, Germon JC, Jayet PA, Soussana JF, Stengel P, editors (2002): *Contribution à la lutte contre l'effet de serre. Stocker du carbone dans les sols agricoles de France?* Expertise Scientifique Collective. Rapport d'expertise réalisé par INRA à la demande du Ministère de l'Ecologie et du Développement Durable. Paris, France: INRA

Audsley E. (coord.), Alber S., Clift R., Cowell S., Crettaz P., Gaillard G., Hausheer J., Jolliet O., Kleijn R., Mortensen B., Pearce D., Roger E., Teulon H., Weidema B., Van Zeijts H. 1997. Harmonisation of Environmental Life Cycle Assessment for Agriculture. Final Report. Concerted Action AIR3-CT94-2028. European Commission. DG VI Agriculture.

Birkved M, Hauschild MZ. 2006. PestLCI—A model for estimating field emissions of pesticides in agricultural LCA. Ecological Modelling **198**(2006) 433–451

Bradley RI, Milne R, Bell J, Lilly A, Jordan C, Higgins A (2005): *A soil carbon and land use database for the United Kingdom*. Soil Use and Management **21**(4): 363-369.

Brandão M, Milà i Canals L, Clift R (submitted) *Soil Organic Carbon changes in the cultivation of energy crops: implications for GHG emissions and soil quality for use in LCA*.

Brimner TA, Gallivan GJ, Stephenson GR. Influence of herbicide-resistant canola on the environmental impact of weed management. *Pest Management Science* 2005; **61**: 47-52.

Bues R, Bussieres P, Dadomo M et al. Assessing the environmental impacts of pesticides used on processing tomato crops. *Agriculture, Ecosystems & Environment* 2004; **102**: 155-62.

Carlsson-Kanyama and Faist. 2000. *Energy Use in the Food Sector: a data survey*. AFN report 291, Swedish Environmental Protection Agency, Stockholm, Sweden.

Carlsson-Kanyama. A. 1998. *Energy consumption and emissions of greenhouse gases in the lifecycle of potatoes, pork meat, rice and yellow peas: methods, data and results from a study of some food consumed in Sweden. With arable land use and re-calculated energy consumption and emissions of greenhouse gases for carrots and tomatoes*. Technical report No. 26, Department of Systems Ecology, Stockholm University, Sweden.

Carlsson-Kanyama A, Boström-Carlsson. 2001. *Energy Use for Cooking and Other Stages in the Life Cycle of Food A study of wheat, spaghetti, pasta, barley, rice, potatoes, couscous and mashed potatoes*. Stockholms Universitet / Systemekologiochfoi. FMS report 160.

Cederberg C, Mattsson B (2000): *Life cycle assessment of milk production — a comparison of conventional and organic farming*. Journal of Cleaner Production, **8**(1) 49-60

Chapin III FS, Woodwell GM, Randerson JT, Rastetter EB, Lovett GM, Baldocchi DD, Clark DA, Harmon ME, Schimel DS, Valentini R, Wirth C, Aber JD, Cole JJ, Goulden ML, Harden JW, Heimann M, Howarth RW, Matson PA, McGuire AD, Melillo JM, Mooney HA, Neff JC, Houghton RA, Pace ML, Ryan MG, Running SW, Sala OE, Schlesinger WH, Schulze E-D (2006): *Reconciling Carbon-cycle Concepts, Terminology and Methods*. Ecosystems **9**:1041-1050

Cowell,S.J. and R.Clift. 1998. "Site-dependency in LCAs involving agricultural production." *8th SETACEurope Annual Meeting*. SETAC-Europe. (April 98 A.D.).

Cross P, Edwards-Jones G. Variation in pesticide hazard from arable crop production in Great Britain from 1992 to 2002: Pesticide risk indices and policy analysis. Crop Protection **25**:1101-1108

DEFRA. 2007. *e-Digest Statistics about: Waste and Recycling*. Department for Environment, Food and Rural Affairs. http://www.defra.gov.uk/environment/statistics/waste/index.htm (accessed 12/10/2007).

Dones R, Faist M, Frischknecht R, Heck T, Jungbluth N. (2004) Life Cycle Inventories of Energy Systems: Results for Current Systems in Switzerland and other UCTE Countries. Final report ecoinvent 2000 No. 5, Paul Scherrer Institut Villigen, Swiss Centre for Life Cycle Inventories, Duebendorf (Switzerland), for ecoinvent members only.

Edwards-Jones G, Koerber G, Milà i Canals L, York EH (2007): *Identifying and managing emissions from farms and food chains*. Carbon Footprint Supply Chain Summit. London (UK) 24-25 May 2007 [http://www.bangor.ac.uk/relu/documents/gejcarbonsummit.ppt]

Fawcett T, Lane K, Boardman B, Banks N, Griffin H, Lipp J, Schiellerup P, Therivel R, Blok K, van Brummelen M, Eising K, Zegers F, Molenbroek E, de Almeida A. T, Nunes C, da Silva J,

Boardman M. 2005. *Lower Carbon Futures for European households*. Environmental Change Institute, University of Oxford.
Finnveden,G. and M.Nilsson. 2005. "Site-dependent Life-Cycle Impact Assessment in Sweden." *International Journal of Life Cycle Assessment*. 10:235-239.
Food Standards Agency (2002) *McCance and Widdowson's The Composition of Foods*, Sixth summary edition. Cambridge: Royal Society of Chemistry. ISBN 0-85404-428-0.
Foster C, Green K, Bleda M, Dewick P, Evans B, Flynn A, Mylan J (2006): *Environmental Impacts of Food Production and Consumption: A report to the Department for Environment, Food, and Rural Affairs*. Manchester Business School. DEFRA, London.
Frischknecht R, Althaus H J, Doka G, Dones R, Heck T, Hellweg S, Hischier R, Jungbluth N, Nemecek T, Rebitzer G, Spielmann M. 2004. *Overview and Methodology*. Final report ecoinvent 2000 No. 1, Swiss Centre for Life Cycle Inventories, Duebendorf, Switzerland. www.ecoinvent.ch
Gallivan GJ, Surgeoner GA, Kovach J. Pesticide risk reduction on crops in the Province of Ontario. *Journal of Environmental Quality* 2001; **30**: 798-813.
Geisler G., Hellweg S., Liechti S., Hungerbuhler K. 2004. Variability Assessment of Groundwater Exposure to Pesticides and its Consideration in Life-Cycle Assessment. Environ. Sci. Technol., 38, 4457-4464
Green K, Foster C. 2005. *Give peas a chance: Transformations in food consumption and production systems*. Technological Forecasting and Social Change **72**(2005):663-679.
Guinée J B, Gorree M, Heijungs R, Huppes G, Kleijn R, Udo de Haes H A, Van der Voet E, Wrisberg M N. 2002. *Life Cycle Assessment. An operational guide to ISO standards.Volume 1, 2, 3*. Centre of Environmental Science, Leiden University (CML), The Netherlands.
Hauschild, M.Z., Thyø K, Jensen KH. 2007. Estimating pesticide emissions for life cycle assessment of agricultural products. SETAC Europe 17th Annual Meeting. Abstracts. Porto (Portugal) 20-24 May 2007.
Herrmann BG, Blok K, Patel MK (2007): *Producing bio-based bulk chemicals using industrial biotechnology saves energy and combats climate change*. Environ. Sci. Technol. **41**(22)7915-7921
Hospido A, Milà i Canals L, McLaren SJ, Clift R, Truninger M, Edwards-Jones G (forthcoming): *Environmental Consequences of Permanent Summertime: exploring the options for year-round lettuce supply to the UK*. In preparation
Huijbregts MAJ, Schöpp W, Verkuijlen E, Heijungs R, Reijnders L (2000): *Spatially explicit characterisation of acidifying and eutrophying air pollution in life-cycle assessment*. Journal of Industrial Ecology **4**:125-142.
International Energy Agency (2007): *Electricity/Heat in Kenya 2004*. On-line: [http://www.iea.org/Textbase/stats/index.asp] Consulted on 26 November 2007
Jarvis, N. 1998. The MACRO Model (Version 4.2). Technical Description; Department of Soil Sciences, Swedish University of Agricultural Sciences: Uppsala, 1998. Cited in Geisler et al. (2004)
Jungmeier G, McDarby F, Evald A, Hohenthal C, Petersen A-K, Schwaiger H-P, Zimmer B (2003): *Energy Aspects in LCA of Forest Products. Guidelines from Cost Action E9*. Int J LCA **8**(2) 99-105
Kim S, Dale BE (2005): *Life cycle assessment study of biopolymers [polyhydroxyalkanoates] derived from no-tilled corn*. Int J LCA **10**(3):200-210
Klein, M. 1995. PELMO - Pesticide Leaching Model. User's Manual. Version 2.01; Fraunhofer-Institut für Umweltchemie und Okotoxikologie: Schmallenberg, 1995. Cited in Geisler et al. (2004)
Koerber G, Milà i Canals L, York EH, Jones DL, Hill PW, Edwards-Jones G (forthcoming): *Integrating soil greenhouse gas emissions into Life Cycle Assessment to improve carbon foot-printing for the food industry*. In preparation
Kovach J, Petzoldt C, Degnil J et al. A method to measure the environmental impact of pesticides. *New York's Food and Life Sci. Bull.* 1992; **139**: 1-8.
Krewitt,W., A.Trukenmüller, T.M.Bachmann, and T.Heck. 2001. "Country-Specific Damage Factors for Air Pollutants. A Step Towards Site-Dependent Life Cycle Impact Assessment." *International Journal of Life Cycle Assessment*. 6:199-210.
Lindeijer,E., R.Müller-Wenk, and B.Steen. 2002. "Impact Assessment of Resources and Land Use." In H.A.Udo de Haes, G.Finnveden, M.Goedkoop, M.Hauschild, E.G.Hertwich, P.Hofstetter, O.Jolliet, W.Klöpffer, W.Krewitt, E.Lindeijer, R.Müller-Wenk, S.I.Olsen, D.W.Pennington, J.Potting, and B.Steen, editors, *Life Cycle Impact Assessment: Striving Towards Best Practice*. SETAC. Pensacola, USA. 11-64.

Margni, M., Rossier, D., Crettaz, P., Jolliet, O., 2002. Life cycle impact assessment of pesticides on human health and ecosystems. Agric. Ecosys. Environ. 93, 379-392.

Maud J, Edwards-Jones G, Quin F. Comparative evaluation of pesticide risk indices for policy development and assessment in the United Kingdom. *Agriculture, Ecosystems and Environment* 2001; **86**: 50-73.

Milà i Canals L. 2003. Contributions to LCA Methodology for Agricultural Systems. Site-dependency and soil degradation impact assessment. 1-250. 2003. PhD Thesis Universitat Autonoma de Barcelona.

Milà i Canals L, Sim S. 2007. *How much carbon for an apple? The carbon footprint of fruit production*. The 64th National Fruit Show 2007. Handbook, pp. 24-27

Milà i Canals L, Domènech X, Rieradevall J, Fullana P, Puig R (2002): *Use of Life Cycle Assessment for the Establishment of the Ecological Criteria for the Catalan Eco-label of Leather*, Int J LCA **7**(1) 39-46

Milà i Canals L, Burnip GM, Cowell SJ (2006) *Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): case study in New Zealand*. Agriculture, Ecosystems & Environment. **114** 226-238

Milà i Canals L, Almudena Hospido, Ivan Muñoz, Sarah J McLaren (2007a): *Life Cycle Assessment (LCA) of Domestic vs. Imported Vegetables. Case studies on broccoli, salad crops and green beans*. CES Working Paper 03/07 Available from http://www.surrey.ac.uk/CES

Milà i Canals L, Bauer C, Depestele J, Dubreuil A, Freiermuth Knuchel R, Gaillard G, Michelsen O, Müller-Wenk R, Rydgren B (2007b): *Key elements in a framework for land use impact assessment in LCA*. Int J LCA **12**(1) 5-15

Milà i Canals L, Romanyà J, Cowell SJ (2007c): *Method for assessing impacts on life support functions (LSF) related to the use of 'fertile land' in Life Cycle Assessment (LCA)*. Journal of Cleaner Production. **15** 1426-1440

Milà i Canals L, Cowell SJ, Sim S, Basson L (2007d): *Comparing Local versus Imported Apples: A Focus on Energy Use*. Env Sci Pollut Res, **14**(5) 338-344

Milà i Canals L, Chenoweth J, Chapagain AK, Orr S, Antón A, Clift R (in preparation a): *Assessing Freshwater Use Impacts in LCA Part I: Inventory Modelling and Characterisation Factors for the Main Impact Pathways*.

Milà i Canals L, Chapagain AK, Orr S, Chenoweth J (in preparation b) *Assessing Freshwater Use Impacts in LCA Part II: Case study for broccoli production in the UK and Spain*.

Milà i Canals L, Hauschild MZ, Cross PA, Edwards-Jones G, Clift R (in preparation c) *Sophisticated pesticide fate modelling: when is it needed for agricultural LCA?*

Muñoz I, Milà i Canals L, Clift R, Doka G (2007): *A Simple Model to Include Human Excretion and Wastewater Treatment in Life Cycle Assessment of Food Products*. CES Working Papers 01/07 Available from http://www.surrey.ac.uk/CES

Muñoz I, Milà i Canals L, Clift R. (submitted). *Consider a spherical man – A simple model to include human excretion in Life Cycle Assessment of Food products*.

Naturvårdsverket. 1997. *Att äta för en bättre miljö. Slutrapport från systemstudie Livsmedel*. Rapport 4830, Stockholm, Sweden. In Swedish.

Nemecek T, Heil A, Huguenin O, Meier S, Erzinger S, Blaser S, Dux D, Zimmermann A (2004): Life Cycle Inventories of Agricultural Production Systems. Final report ecoinvent 2000 No. 15. Agroscope FAL Reckenholz and FAT Taenikon, Swiss Centre for Life Cycle Inventories, Dübendorf (Switzerland)

NFA, National Food Administration. 1985. *Svinnet i livsmedelshanteringen*. In Swedish.

Nguyen TLT and Gheewala SH (in press): *Life Cycle Assessment of Fuel Ethanol from Cassava in Thailand*, Int J LCA [DOI: http://dx.doi.org/10.1065/lca2007.06.343]

Nebel B, Cowell SJ (2003): *Global Warming Reduction Potential of biomass based products: An example of wood products*. SETAC 2003 European Annual Meeting. Abstracts

Nebel B, Zimmer B, Wegener G (2006): *Life Cycle Assessment of Wood Floor Coverings – A Representative Study for the German Flooring Industry*. Int J LCA **11** (3) 172 – 182

Nielsen P. 2003. *Cool and frozen storage in wholesale*. http://www.lcafood.dk/. Accessed 9 august 2007.

Notarnicola, B., Tassielli, G., Nicoletti, G.M., 2003., LCC and LCA of extra-virgin olive oil: organic vs. conventional. In: Halberg, N. (Ed.), Life Cycle Assessment in the Agri-food sector. Proceedings from the 4th International Conference, October 6-8, 2003, Bygholm, Denmark. DIAS Report no. 61, Tjele, Denmark, pp. 58-63.

Öko-Institut. 2000. *Bus – city – Euro 1*. Dataset included in the GEMIS software database 4.3. http://www.oeko.de/service/gemis/en/index.htm

Olsson P. LCAnet Food. Final Document. 1999. Gothenburg, Sweden, SIK.
Owens JW. 2002. *Water Resources in Life-Cycle Impact Assessment. Considerations in Choosing Category Indicators*. Journal of Industrial Ecology **5**:37-54.
Piringer G, Steinberg LJ (2006): *Reevaluation of Energy Use in Wheat Production in the United States*. Journal of Industrial Ecology **10**:149-167
Potting J, W Schöpp, K Blok, and M Hauschild. 1998. "Site-dependent life cycle impact assessment of acidification." *Journal of Industrial Ecology*. 2:63-87.
Pretty J. N, Ball A.S, Lang T, Morison J. I. L. 2005. *Farm costs and food miles: An assessment of the full cost of the UK weekly food basket*. Food Policy 30, 1–19.
Rabl A, Benoist A, Dron D, Peuportier B, Spadaro JV, Zoughaib A (2006): *How to Account for CO2 Emissions from Biomass in an LCA*. Int J LCA **12** (5) 281
Ritchie, K. 2005. *From farm to table: An energy consumption assessment of refrigerated, frozen and canned food delivery*. Draft version. A report prepared by Scientific Certification Systems Inc. on behalf of the Steel Recycling Institute, Pennsylvania, United States.
Ross S, D Evans. 2002. "Excluding Site-Specific Data from the LCA Inventory: How This Affects Life Cycle Impact Assessment." *International Journal of Life Cycle Assessment*. 7:141-150.
Satish SV, Ranganna B. 2002. *Effect of curing on the physical and mechanical properties of onion*. Mysore Journal of Agricultural Sciences **36** (3) 266-269
Smith DT, Harris MK, Liu TX. Adoption of pest management practices by vegetable growers: A case study. *American Entomologist* 2002; **48**: 236-42.
Sonesson,U., H.Janestad, and B.Raaholt. *Energy for Preparation and Storing of Food - Models for calculation of energy use for cooking and cold storage in households*. 709 2003, 1-56. 2003. Gothenburg, Sweden, SIK. SIK-Rapport.
Spielmann M, Kägi T, Tietje O. (2004) Life Cycle Inventories of Transport Services. Final report ecoinvent 2000 No. 14, Swiss Centre for Life Cycle Inventories, Duebendorf (Switzerland), for ecoinvent members only
The composting association. 2006. *The State of Composting and Biological Waste Treatment in the UK. 2004/05*. Published by The Composting Association. Supported by Environment Agency and WRAP. Northamptonshire, England.
Weidema B.P, Pedersen R.L, and Drivsholm T.S. 1996. *Lifecycle screening of food products – two examples and some methodological proposals*. Danish Academy of Technical Sciences (ATV).
Wenzel,H. 1998. "Application Dependency of LCA Methodology: Key Variables and Their Mode of Influencing the Method." *International Journal of Life Cycle Assessment*. 3:281-288.
Woomer PL, Plalm CA, Qureshia JN, Kotto-Same J (1997): *Carbon sequestration and organic resources management in African smallholder agriculture*. In: Lal R, Kimble JM, Follett RF,
Stewart BA, editors. Management of carbon sequestration in soil. Boca Raton, USA: CRC Press; p. 153-173
WRAP (2007): Understanding Food Waste. Research Summary. Available from www.wrap.org.uk

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