10

Uncertainty Analysis of the Life-Cycle Greenhouse Gas Emissions and Energy Renewability of Biofuels

João Malça[1,2] and Fausto Freire[1]
[1]*ADAI-LAETA, Dept of Mech. Engineering, University of Coimbra, Coimbra,*
[2]*Dept of Mech. Engineering, ISEC, Coimbra Polytechnic Institute, Coimbra*
Portugal

1. Introduction

Biofuels can contribute substantially to energy security and socio-economic development. However, significant disagreement and controversies exist regarding the actual energy and greenhouse gas (GHG) savings of biofuels displacing fossil fuels. A large number of publications that analyze the life-cycle of biofuel systems present varying and sometimes contradictory conclusions, even for the same biofuel type (Farrell et al., 2006; Malça and Freire, 2004, 2006, 2011; Gnansounou et al., 2009; van der Voet et al., 2010; Börjesson and Tufvesson, 2011). Several aspects have been found to affect the calculation of energy and GHG savings, namely land use change issues and modeling assumptions (Gnansounou et al., 2009; Malça and Freire, 2011). Growing concerns in recent years that the production of biofuels might not respect minimum sustainability requirements led to the publication of Directive 2009/28/EC in the European Union (EPC 2009) and the National Renewable Fuel Standard Program in the USA (EPA 2010), imposing for example the attainment of minimum GHG savings compared to fossil fuels displaced.
The calculation of life cycle GHG emission savings is subject to significant uncertainty, but current biofuel life-cycle studies do not usually consider uncertainty. Most often, life-cycle assessment (LCA) practitioners build deterministic models to approximate real systems and thus fail to capture the uncertainty inherent in LCA (Lloyd and Ries, 2007). This type of approach results in outcomes that may be erroneously interpreted, or worse, may promote decisions in the wrong direction (Lloyd and Ries, 2007; Plevin, 2010). It is, therefore, important for sound decision support that uncertainty is taken into account in the life-cycle modeling of biofuels. Under this context, this chapter has two main goals: i) to present a robust framework to incorporate uncertainty in the life-cycle modeling of biofuel systems; and ii) to describe the application of this framework to vegetable oil fuel in Europe. In addition, results are compared with conventional (fossil) fuels to evaluate potential savings achieved through displacement. Following this approach, both the overall uncertainty and the relative importance of the different types of uncertainty can be assessed. Moreover, the relevance of addressing uncertainty issues in biofuels life-cycle studies instead of using average deterministic approaches can be evaluated, namely through identification of important aspects that deserve further study to reduce the overall uncertainty of the system.

This chapter is organized in four sections, including this introduction. Section 2 presents the comprehensive framework developed to capture uncertainty in the life-cycle GHG emissions and energy renewability assessment of biofuels, addressing several sources of uncertainty (namely parameter and modeling choices). Section 3 describes and discusses the application of this framework to vegetable oil fuel in Europe. Section 4 draws the conclusions together.

2. Framework: Energy and GHG life-cycle modeling addressing uncertainty

This section presents the biofuel life-cycle modeling framework used in this chapter. The most relevant methodological issues and sources of uncertainty in the energy and GHG assessment of biofuels are also discussed.

2.1 Life-cycle assessment of biofuels

A Life-Cycle Assessment (LCA) study offers a comprehensive picture of the flows of energy and materials through a system and gives a holistic and objective basis for comparison. The LCA methodology is based on systems analysis, treating the product process chain as a sequence of sub-systems that exchange inputs and outputs. The results of an LCA quantify the potential environmental impacts of a product system over the life-cycle, help to identify opportunities for improvement and indicate more sustainable options where a comparison is made. The LCA methodology consists of four major steps (ISO 14044, 2006):

- The first component of an LCA is the definition of the goal and scope of the analysis. This includes the definition of a reference unit, to which all the inputs and outputs are related. This is called the functional unit, which provides a clear, full and definitive description of the product or service being investigated, enabling subsequent results to be interpreted correctly and compared with other results in a meaningful manner;
- The second component of an LCA is the inventory analysis, also Life-Cycle Inventory (LCI), which is based primarily on systems analysis treating the process chain as a sequence of sub-systems that exchange inputs and outputs. Hence, in LCI the product system (or product systems if there is more than one alternative) is defined, which includes setting the system boundaries (between economy and environment, and with other product systems), designing the flow diagrams with unit processes, collecting the data for each of these processes, leading with multifunctional processes and completing the final calculations. Its main result is an inventory table, in which the material and energy flows associated with the functional unit are compiled and quantified;
- The third component of an LCA is the Life-Cycle Impact Assessment (LCIA), in which the LCI input and output flows are translated into potential contributions to environmental impacts. Different methods and models are available to conduct this step, based on aggregating and reducing the large amount of LCI data into a limited number of impact categories;
- Finally, interpretation is the fourth component of an LCA. The results of the life-cycle study are analyzed, so that conclusions can be drawn and recommendations made, according to the scope and objectives of the study.

Life-cycle studies of biofuel systems can be classified into three groups (Liska and Cassman, 2008; Cherubini and Strømman, 2011):

- life-cycle energy analysis, focused on fossil fuel requirements, energy efficiency and/or characterizing biofuel renewability);

- life-cycle GHG assessment (calculating the GHG balance); and
- life-cycle assessment, in which a set of environmental impact categories are investigated.

Furthermore, concerning the particular purpose of the biofuel LCA studies, the following subdivision can be made (van der Voet et al., 2010):

- comparative LCA, in which biofuel systems are compared with their fossil fuel equivalents on a life-cycle basis (e.g. GHG calculators used by governments to support biofuel policies);
- biofuel LCA used to obtain insight into the main environmental impacts of a specific chain (e.g. for generation of data on new production processes); and
- biofuel LCA used to identify main hotspots in the chain, which are specially suited for biofuel production companies aiming at realizing improvements in their processes.

Important methodological challenges within the field of biofuel LCA can be identified, namely concerning the choice of functional unit and definition of system boundaries. The definition of a functional unit is an important step in a Life-Cycle Assessment (Cherubini, 2010): it is a quantified description of the identified functions (performance characteristics) of a product system and provides a reference to which all other data (inputs and outputs) in the assessment are related (ISO 14040, 2006). The definition of the functional unit in biofuel life-cycle studies is related to the scope and system boundaries of the study; therefore, there is no single or preferred functional unit for biofuel assessments. The most common functional units found in the literature are (van der Voet et al., 2010; Malça and Freire, 2011):

- Service-oriented, e.g. 1 km driven in a specific vehicle;
- Energy-oriented, e.g. 1 MJ of biofuel energy content;
- Mass-oriented, e.g. 1 kg of biofuel produced;
- Volume-oriented, e.g. 1 liter of biofuel produced; and
- Land area-oriented, e.g. 1 ha of land for energy crop production.

The option for mass- or volume-based functional units have been used in several studies (e.g. Shapouri et al., 1995; Kim and Dale, 2002; Shapouri et al., 2002). However, in most cases this is not an adequate basis for comparison of the function provided by different (bio)fuels.

The functional unit chosen for the application reported in this chapter is 1 MJ of the final (bio)fuel product, measured in terms of the lower heating value (LHV, heat of combustion excluding the latent heat in combustion products, i.e. the specific enthalpy of vaporization of water). This functional unit is consistent with the goal and scope, which is to calculate the life-cycle GHG intensity (g CO_2eq MJ^{-1}) and energy renewability efficiency of European rapeseed oil and compare these values with their fossil fuel equivalents. Therefore, the system has been modeled taking into account the energy and GHG emissions required to deliver the biofuel to the end user, namely biomass cultivation, processing, transportation and storage of raw materials, followed by biofuel production and distribution. Setting theses boundaries is appropriate, because the goal and scope is concerned with biofuel use as a generic energy carrier, without a particular transportation or energy conversion system being considered. This assessment enables life-cycle inventory results to be analyzed in a variety of different ways, including hotspot identification and optimization of the biofuel chain, as well as calculation of potential energy and GHG reductions over fossil fuels.

Calculation of energy and GHG savings of biofuel systems requires the establishment of an appropriate baseline. The definition of a reference system is particularly used by legislation, which sets minimum levels for GHG emission savings that biofuels must achieve (e.g. EPC, 2009; EPA, 2010). Most commonly, the reference system used is a fossil fuel pathway (gasoline or diesel). However, the EU directive 2009/28/EC (EPC, 2009) has adopted a generic reference value for fossil fuels used for transportation (83.8 g CO_2eq MJ^{-1}), not distinguishing between petrol and diesel. For bioliquids used for electricity production the reference value adopted is 91 g CO_2eq MJ^{-1}, for bioliquids used for heat production the value is 77 g CO_2eq MJ^{-1}, and for cogeneration is 85 g CO_2eq MJ^{-1}. A justification for adopting distinct values based on the type of final use and not on the fossil fuel displaced could not be found in directive 2009/28/EC. In this chapter, petroleum diesel is the reference system, and includes extraction, transport and refining of crude oil, and distribution of final fuel.

2.2 Energy analysis

Several reasons motivate the sometimes diverging results of life-cycle energy analyses of biofuel systems, namely (i) the quantification of energy fluxes either in terms of final energy or in terms of primary energy; and (ii) the use of different metrics for energy efficiency. These topics are explored in this section.

Energy resource depletion must be quantified in terms of primary energy – energy embodied in natural resources (e.g. coal, crude oil, uranium or biomass) that has not undergone any anthropogenic conversion or transformation. Primary energy is the sum of the final energy with all the transformation losses, with fuel primary energy values being greater than their final energy values. In fact, consumers buy final energy, but what is really consumed is primary energy, which represents the cumulative energy content of all resources (renewable and non-renewable) extracted from the environment. In the case of fuels, energy inputs required during the extraction, transportation and production processes measured in terms of primary energy ($E_{in,prim}$, MJ kg^{-1}), do not include the energy embodied in the final fuel, i.e. the fuel energy content (FEC, MJ kg^{-1}). Even though, the energy requirement of fossil fuels should also include the FEC, in which case the result is referred to as the gross energy requirement (GER, MJ kg^{-1}) (Mortimer et al., 2003):

$$GER = E_{in,non\text{-}renewable,prim} + FEC \tag{1}$$

In (bio)energy analysis studies it is essential to distinguish between non-renewable ($E_{in,non\text{-}renewable,prim}$) and renewable ($E_{in,renewable,prim}$) energy inputs, because we are concerned with the renewable nature of biofuels and the depletion of fossil fuels. Therefore, the essential comparison that needs to be made is between the non-renewable primary energy input to the biofuel life-cycle ($E_{in,non\text{-}renewable,prim}$) and the non-renewable primary energy requirements throughout the life-cycle of fossil fuels, including the fossil fuel energy content, i.e. the GER.

The life-cycle inventory results provide an opportunity to quantify the total energy demand and, therefore, the overall energy efficiency. Quantifying the overall energy efficiency of a biofuel is helpful to determine how much (non-renewable) energy must be expended to produce biomass and convert its energy to 1 MJ of available energy in the transportation fuel. The more non-renewable energy is required to make the biofuel, the less we can say the biofuel is “renewable”. Thus, the renewable nature of a fuel can vary across the spectrum of “completely renewable” (i.e. zero non-renewable energy inputs) to non-renewable (i.e. non-renewable energy inputs as much or more than the energy output of the fuel) (Sheehan et al., 1998).

Within the energy analysis and LCA literature there is lack of consensus concerning the definition (and designation) of energy efficiency indicators to be used in a life-cycle perspective and, in particular, to characterize the energy requirements of renewable energy systems. In fact, various indicators have been used, often with the same meaning but different definition, or inversely, e.g. overall energy efficiency (Boustead and Hancock, 1979; Boustead, 2003); energy efficiency (ADEME, 2002); gross energy requirement and net energy requirement (Wilting 1996); energy requirement (Whitaker et al., 2010); overall energy balance (Armstrong et al., 2002); energy balance (Börjesson and Tufvesson, 2011); cumulative energy demand (Huijbregts et al., 2006); input/output energy balance, cumulative energy requirement, fossil energy requirement, and renewable energy requirement (Cherubini et al., 2009); net energy use, and energy substitution efficiency (Gnansounou et al., 2009); energy ratio (Liska and Cassman, 2008; Papong and Malakul, 2010); net energy yield (Liska and Cassman 2008); and energy return on investment[1] (Poldy, 2008). In particular, Sheehan et al. (1998) have used the life-cycle energy efficiency (LCEE), defined as the ratio between the biofuel energy content and the biofuel GER:

$$LCEE = \frac{\mathrm{FEC}}{(\mathrm{E}_{\mathrm{in,\ non-renewable,prim}} + \mathrm{FEC})} \tag{2}$$

The LCEE can be seen as a measure of the fraction of the GER (primary energy required throughout the biofuel life-cycle plus the biofuel energy content), which actually ends up in the fuel product. The same authors (and others, e.g. Lechón et al., 2009) have also adopted the fossil energy ratio (FER), defined as:

$$FER = \frac{\mathrm{FEC}}{E_{in,non-renewable,prim}} \tag{3}$$

According to this definition, if the fossil energy ratio is less than 1 the fuel is nonrenewable, as more energy is required to make the fuel than the energy available in the final fuel product. Biofuel with FER greater than 1 can be considered as (partially) renewable. In theory, a total renewable fuel would have no fossil energy requirement and, thus, its fossil energy ratio would be infinite. Other authors have also used the FER indicator, but under a different designation, for example "energy efficiency" (ADEME, 2002), whereas others have used the "energy requirement" (E_{req}), defined as the "primary energy input per delivered energy output" (Mortimer et al., 2003; Malça and Freire, 2004, 2006; Hoefnagels et al., 2010):

$$E_{req} = \frac{E_{in,non-renewable,prim}}{\mathrm{FEC}} \tag{4}$$

The energy requirement indicator is also used in Kim and Dale (2002) and Armstrong et al. (2002), but under the designation of "net energy" and "overall energy balance", respectively. It should be noted that E_{req} is the inverse of FER.
The "net energy value" (NEV), defined as the biofuel FEC minus the non-renewable energy required to produce the biofuel):

[1] To distinguish it from a financial measure, the energy return on investment (EROI) is sometimes called energy return on energy investment (EROEI) (Poldy, 2008).

$$NEV = FEC - E_{in,non-renewable,prim} \tag{5}$$

is used e.g. in Shapouri et al. (1995), Shapouri et al. (2002), Liska and Cassman (2008) and Papong and Malakul (2010)[2]. In this case, negative net energy values indicate that (bio)fuel is non-renewable, while positive values indicate the fuel is renewable to a certain extent.

According to Liska and Cassman (2008) and Cherubini et al. (2009), input–output ratios and primary energy requirements receive most attention when assessing the efficiency of bioenergy systems, because they provide a straightforward basis for comparison with conventional fossil fuel systems. Moreover, these metrics are usually thought as a surrogate for GHG emissions mitigation (Liska and Cassman, 2008). Nevertheless, intensity factors do not provide a measure of the "energy productivity" of a system on a land-area basis, which should be the chosen parameter when dedicated energy crops compete with food, feed or fiber under land-availability constraints (Liska and Cassman, 2008; Cherubini et al., 2009; Cherubini and Strømman, 2011). An example is the net energy yield NEY (GJ ha^{-1}) used by Liska and Cassman (2008), which combines energy efficiency and productivity into one single parameter.

Another metric, the Energy Renewability Efficiency, aiming at characterizing the renewability of (bio)fuel systems has been proposed by Malça and Freire (2004, 2006). The energy renewability efficiency (ERenEf) measures the fraction of final fuel energy obtained from renewable sources by subtracting from FEC all the inputs of non-renewable primary energy (Malça and Freire, 2006). It thus provides a more adequate means for quantifying the renewability degree (or its lack) of a particular energy system. ERenEf can be defined as:

$$ERenEF[\%] = \frac{\left(\mathrm{FEC} - \mathrm{E}_{\mathrm{non-renewable,prim}}\right)}{\mathrm{FEC}} \times 100 \tag{6}$$

A biofuel may be considered renewable if ERenEf assumes values between 0 and 100%. In case there were no inputs of non-renewable energy, the biofuel would be completely renewable with an ERenEf of 100%. If the ERenEf is lower than zero, then the biofuel should be characterized as non-renewable since the non-renewable energy required to grow and convert biomass into biofuel would be greater than the energy present in the biofuel final product. In this case, the biofuel is, indeed, not a fossil energy substitute and increasing its production does little to displace oil imports or increase the security of energy supply. By definition, non-renewable energy sources have negative values of ERenEf, with increasing negative values as life-cycle energy efficiency decreases. For example, fossil diesel (the fossil fuel displaced by rapeseed oil shows an average ERenEf value of –14.0%, meaning that the total primary energy required to produce fossil diesel is 14.0% greater than its final energy content.

2.3 GHG assessment

This section presents the methodology used for calculating the GHG balance of biofuel systems. Important issues in the GHG assessment of biofuels, such us carbon stock changes associated with land use change and soil emissions from land use, and how they are

[2] Papong and Malakul (2010) also use this net energy definition, although under the name "Net Energy Gain".

addressed in the practical modeling of the life-cycle are discussed. Generic assumptions concerning GHG accounting are also formulated.

The life-cycle GHG balance of biofuel systems can be calculated by summing up the GHG emissions of the several process steps, namely land use change, cultivation of raw materials (soil preparation, fertilization, sowing, weed control, and harvesting) and biofuel production (transport, storage and drying of feedstock, processing of feedstock into biofuel, and biofuel transport to the final user). Biofuel use (combustion in engines or boilers) is not explicitly modeled, but is assumed that tailpipe CO_2 emissions from biofuel combustion are neutral, being balanced by the CO_2 sequestered during crop growth, which does not occur for fossil fuels. An alternative approach would be to distinguish between fossil and biogenic CO_2 emissions throughout the life-cycle (see e.g. Rabl et al. 2007; Guinée et al. 2009; Luo et al. 2009).

One emerging but highly controversial issue in the GHG balance of biofuels is indirect land use change (iLUC) (Anex and Lifset, 2009; Liska and Perrin, 2009). In the approach proposed in this chapter iLUC is not considered, but a brief discussion is presented at the end of this section. The greenhouse gases considered are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), with average global warming potentials (100 year time horizon) of $GWP_{CH4}=25$ and $GWP_{N2O}=298$. Other GHG emissions from biofuel systems were found to be negligible and were not pursued. Global Warming Potentials used by the IPCC provide "CO_2 equivalence" factors for greenhouse gases other than CO_2, which allows aggregation of emissions of different gases into a single metric (IPCC, 2007). In terms of global warming, $GWP_{CH4}=25$ means that 1 g of methane released to the atmosphere is equivalent to the release of 25 g of carbon dioxide. In practical terms, GHG emissions in each step are multiplied by the respective equivalence factors and summed up yielding a single figure in CO_2 equivalents. Finally, the GHG emissions of the overall biofuel chain can be calculated.

GHG emissions for feedstock and energy inputs are calculated by using suitable emission factors (Mortimer and Elsayed, 2006; Malça and Freire, 2010).

For comparative and decision purposes, GHG emission savings can be calculated by comparing the life-cycle GHG emissions of biofuels with the equivalent emissions of fossil fuels, following the methodology used e.g. in EPC (2009):

$$GHG_{emission\ savings}\ [\%] = \frac{(\text{Fossil Fuel}_{\text{emissions}} - \text{Biofuel}_{\text{emissions}})}{\text{Fossil Fuel}_{\text{emissions}}} \times 100 \quad (7)$$

DIRECT LAND USE CHANGE AND LAND USE. Soil carbon stock change is an emergent topic in the literature and can contribute significantly to biofuel GHG intensity (EC, 2010a). However, it is site specific and highly dependent on former and current agricultural practices, climate and soil characteristics and, thus, previous biofuel LCA studies have neglected this issue (Larson, 2006; Malça and Freire, 2011). A change in land use (for example, set-aside land to cropland) or in agronomic practices (change to low tilling, for example) can liberate carbon that had previously been sequestered over a long period of time or, conversely, lead to a carbon build-up in the soil (Cherubini and Strømman, 2011). Moreover, soil organic carbon (SOC) stock exchange is a relatively slow process and thus difficult to measure (Heller et al., 2003). IPCC (2006) guidelines indicate a default time period for transition between equilibrium SOC values (i.e. soil carbon levels from which there is no further net accumulation or degradation) of 20 years.

Annualized soil carbon stock variations due to land use change and practices ΔC_{LUC-a} (tonnes per hectare per year, t C ha^{-1} yr^{-1}) are given by (EPC, 2009)

$$\Delta C_{LUC-a} = \frac{CS_R - CS_A}{T_{LUC}} \tag{8}$$

in which CS_R (tC ha^{-1}) is the carbon stock (CS) per unit area of the reference land use (cropland, set-aside land or grassland), CS_A (tC ha^{-1}) is the carbon stock per unit area associated with the arable use of soils, and T_{LUC} (yr) is the time period for transition between equilibrium carbon stocks. Actually, set-aside lands and grasslands placed in cultivation lose soil carbon at an exponential rate (JEC, 2007): most of the carbon loss occurs within the first few years following initial cultivation. A discussion of the temporal dynamics of GHG emissions caused by land use change is, however, beyond the scope of this chapter.

Carbon stocks per unit area CS_R and CS_A include both soil and vegetation and can be calculated according to EC (2010) rules. The soil organic carbon (SOC) content is given by SOC = $SOC_{ST}.F_{LU}.F_{MG}.F_I$, in which SOC_{ST} is the standard soil organic carbon in the 0-30 cm topsoil layer, F_{LU} is a factor reflecting the type of land use, F_{MG} reflects the adopted soil management practices and F_I quantifies the level of carbon input to soil. Carbon stock values concerning above and below ground vegetation as provided in EC (2010) guidelines are also included in the calculation of the overall land carbon stock.

Several authors call the amount of CO_2 emissions from land use change the "carbon debt" of land conversion (Fargione et al., 2008). Over time, this carbon debt can be gradually compensated if GHG emission savings of growing biofuels while displacing fossil fuels are realized. The period of time that biofuel production takes to repay the carbon debt is called the carbon payback time; it is calculated by dividing the net carbon loss from LUC per hectare by the amount of carbon saved per hectare and per year by the use of biofuels, excluding LUC emissions (Wicke et al., 2008).

The calculation of GHG emissions also includes emissions of nitrous oxide (N_2O) from soil. The assessment of N_2O emissions from soil has recently proven to be an important issue in the GHG balance of biofuels (Crutzen et al., 2008; Reijnders and Huijbregts, 2008). Agricultural practices, and particularly the use of fertilizers containing nitrogen, are important issues affecting the emission of N_2O from soils (Kaiser et al., 1998; Reijnders and Huijbregts, 2008). Generally, a small amount of the nitrogen in the fertilizer ends up being released to the atmosphere as N_2O, both *i)* directly, from nitrification of nitrogen in the fertilizer and from crop residues; and *ii)* indirectly, following volatilization of NH_3 and NO_x and after leaching and runoff of N from managed soils (IPCC, 2006). Because N_2O has a high impact on global warming, its emissions from agricultural soils cannot be neglected. The contribution to net emissions of N_2O from nitrogen fertilizer application is one of the most uncertain variables due to the number of parameters that can affect its value (Larson, 2006). Actual emissions from fields vary depending on soil type, climate, tillage method, fertilizer application rates and crop type (Larson, 2006; Reijnders and Huijbregts, 2008; Stephenson et al., 2008; Crutzen et al., 2008).

INDIRECT LAND USE CHANGE. An aspect that requires a consequential approach in life-cycle studies is the assessment of indirect land use change associated with biofuels. Increased biofuel demand may lead to an expansion of cropped area at the expenses of other land uses. The displacement of prior crop production to other areas (indirect LUC) may contribute to important environmental impacts, namely GHG emissions (Fargione et al., 2008; Searchinger et al., 2008; Wicke et al., 2008), which has recently been the subject of important controversy among the scientific community. This builds on the fact that market mechanisms should be taken into account when modeling all the consequences of increased consumption of biofuels, which requires subjective assumptions and leads to potentially higher complexity and uncertainty.

A report by Croezen et al. (2010) discussed the use of different agro-economic models – simulating global agricultural markets, trade, intensification, possible crop replacements – to estimate iLUC implications and showed that overall emissions from iLUC are within 10 to 80 g CO_2 MJ^{-1} of biofuel produced. Other attempts for addressing indirect land use change and its influence on life-cycle results, namely through the use of single CO_2 emission factors –the iLUC factor approach–, have also been conducted (e.g. Bowyer, 2010; Fritsche et al., 2010). Nevertheless, these models likely estimate GHG emissions from iLUC with significant inaccuracy (Cherubini and Strømman, 2011). Further work is still required to address the practical modeling of indirect LUC associated with biofuels, as stated e.g. by Anex and Lifset (2009), Liska and Perrin (2009) and Kløverpris et al. (2008), so that a harmonized methodology can be established. Also, the EU recognizes in a report published on December 2010 (EC, 2010b) that a number of uncertainties associated with iLUC modeling remain to be addressed, which could significantly impact the results. Therefore, indirect LUC is beyond the scope of this chapter.

2.4 Uncertainty analysis

Uncertainty analysis is a systematic procedure to determine how uncertainties in data and assumptions propagate throughout a life-cycle model and how they affect the reliability of the life-cycle study outcomes. Uncertainties may occur in the several phases of an LCA, namely in the goal and scope definition, inventory analysis and impact assessment. Examples are provided e.g. in Björklund (2002), Huijbregts (1998), Heijungs and Huijbregts (2004), and Geisler et al. (2005).

In general, results of a life-cycle study can be uncertain for a variety of reasons (Morgan and Henrion, 1990; Huijbregts, 1998; Björklund, 2002; Huijbregts et al., 2003; Heijungs and Huijbregts, 2004; Lloyd and Ries, 2007), and different typologies can be used to describe the uncertainties considered. According to Huijbregts (1998), the following sources of uncertainty in LCA can be distinguished:

- parameter uncertainty, which arises from lack of data, empirical inaccuracy (imprecise measurements), and unrepresentativity of data (incomplete or outdated measurements);
- uncertainty due to choices (or scenario uncertainty), which reflects the inherent dependence of outcomes on normative choices in the modeling procedure (e.g. choice of functional unit, definition of system boundaries, or selection of allocation methods); and
- model uncertainty, due to the use of mathematical relationships between model inputs and outputs that simplify real-world systems.

In general, parameter and model uncertainty are characterized by means of probability distributions, whereas uncertainty due to choices is addressed through the development of unique scenarios (Lloyd and Ries, 2007; Malça and Freire, 2010).

PARAMETER UNCERTAINTY. Every type of modeling is associated with uncertainties in its parameters (Schade and Wiesenthal, 2011). In this article, a robust approach is used to address and incorporate parameter uncertainty in the life-cycle modeling of rapeseed oil. The main steps of this approach can be summarized as follows:

- firstly, a preliminary sensitivity analysis is conducted, in which single parameter variations are tested to see how the results are affected. The merit of this step is to identify the parameters with the highest impact on the model outputs, and thus the parameters that require particular attention in the next steps;
- secondly, a literature review is conducted to identify variation ranges and assign appropriate probability density functions for the most influential parameters;
- thirdly, an uncertainty propagation method is used (with Monte-Carlo simulation) for calculating probability distributions of output variables based on the uncertainty within selected input parameters;
- finally, an uncertainty importance analysis is conducted in order to identify the parameters that contribute most to the overall output variance.

Although widely used, single sensitivity analysis generally underestimates the uncertainty in a model (Plevin, 2010), as e.g. with non-linear models, where the sensitivity to a specific parameter depends on the nominal values assigned to other variables (Saltelli et al., 2006). This case requires that sensitivity is assessed with parameters varying simultaneously, i.e. using global sensitivity analysis. A common technique for global sensitivity analysis is Monte-Carlo simulation. Monte-Carlo simulation is based on the repetition of many individual model iterations (typically from hundreds to thousands), with each iteration using a randomly constructed set of values selected from each parameter probability distribution. The set of model outputs computed by the simulation is then aggregated into a probability distribution. The Oracle Crystal Ball software package was used to perform Monte-Carlo simulation (Oracle, 2010).

To compare the relative importance of the uncertainty in input parameters to the model output uncertainty, an uncertainty importance analysis is performed. Generally, a limited number of parameters account for the majority of uncertainty in the model outputs (Morgan and Henrion, 1990). The merit of estimating uncertainty importance is to identify these parameters, and thus guide further research to reduce their uncertainty. Moreover, the remaining parameters (typically a much larger set), which contribute negligibly to the overall variance, can be treated as uncertain, simplifying the model and saving computation time.

UNCERTAINTY OF GLOBAL WARMING POTENTIALS. Several time horizons can be adopted for the estimation of GHG emissions. Taking into account the short- to mid-term implications of first generation biofuels in terms of global warming effect, the most commonly used time horizon of 100-years has been chosen for GWP estimation in the application presented in section 3. Nonetheless, other time horizons can be adopted. Results with GHG emissions for various time horizons (20, 100 and 500-year) have been calculated by the authors of this chapter and it has been concluded that 500-yr GHG emissions are lower due to a significantly lower GWP of nitrous oxide (153 vs. 298 kg CO_2eq for 500- and 100-yrs, respectively). Moreover, uncertainty ranges for a 500-yr timeframe are narrower than corresponding 100-year values, because of the lower uncertainty in the estimation of GWP_{N2O}. On the other hand, calculated RO GHG emissions for 20- and 100-yr time horizons are similar, because 20- and 100-yr GWPs of N_2O are also very similar. Since methane (CH_4) hardly contributes to the life-cycle GHG emissions of RO (Malça and Freire, 2009), the implications of GWP_{CH4} variation between different time horizons are not significant. An uncertainty of ±35% for the 90% confidence range has been considered for GWP_{CH4} and GWP_{N2O}, according to IPCC (2007).

MULTIFUNCTIONALITY (Scenario Uncertainty). Most industrial and agricultural processes are multifunctional. In particular, many of the feedstocks for biofuels are either co-produced with other products or are from by-products from other production processes. Biofuel production systems generate large quantities of co(by)-products and thus LCA practitioners are faced with the problem that the product system under study provides more functions than that which is investigated in the functional unit of interest. This leads to the following central question: how should the resource consumption and energy used be distributed over the various co(by)-products? An appropriate procedure is required to partition the relevant inputs and outputs to the functional unit under study.

The international standards on LCA include several options for dealing with co-production (ISO 14044, 2006): i) sub-dividing the process into two or more sub-processes; ii) expanding the product system to take into account potential effects of providing a new use for the co-products on systems currently using the co-products – known as system boundary expansion – and iii) allocating inputs and outputs between product streams based on causal relationships.

Although allocation methods are straightforward to implement, they "arbitrarily" allocate inputs and outputs on the basis of specific relationships between co-products (Weidema, 2003). For this reason, ISO standards on LCA indicate that allocation[3] should be avoided, wherever possible, in favor of subdividing the system in sub-processes (often not possible) or by expanding the system (system boundary expansion). As explained by Guinée et al. (2009), system expansion (also called system extension) means extending the product system to include additional functions related to the co-products. As a result, the system includes more than one functional unit. Sometimes the expression "system extension" refers to what actually is the "substitution method" (also called "replacement method", "displacement method" or "avoided-burdens" approach). Substitution refers to expanding the product system with "avoided" processes to remove additional functions related to the functional flows of the system. In this case, energy and emission credits can be assumed equal to those required to produce a substitute for the co-products.

Allocation can be based on physical properties of the products, such as mass, volume, energy, carbon content, because data on the properties are generally available and easily interpreted. Where such physical causal relationships cannot be used as the basis for allocation, the allocation should reflect other relationships between the environmental burdens and the functions. Many biofuel life-cycle studies use the mass of co-products as the basis for partitioning the system (e.g. ADEME, 2002; Neupane et al., 2011). Other studies use the energy content (e.g. Janulis, 2004; Wagner et al., 2006). However, the main reason for using mass seems to arise because both main and co-products can be weighted, and the use of energy content would only be relevant if both main and co-products were actually burned as fuels. Nonetheless, mass and energy allocation factors do not change over time, like economic factors or substituted product types do (Hoefnagels et al., 2010). At the European policy level, energy allocation has been selected as the method for the regulation of individual economic operators, because it is easy to apply, is predictable over time and

[3] The meaning of allocation in LCA is often used misleading. According to ISO 14044:2006, sub-division and system boundary expansion are not formally part of the allocation procedure.

minimizes counter-productive incentives (EPC, 2009). Allocation can also be based on the exergy (e.g. Frischknecht, 2000; Dewulf et al., 2005) or carbon (e.g. Gnansounou et al., 2009) content of the co-products. Allocation based on the relative economic value (market price) of main and co-products is used e.g. by Guinée et al. (2004), Zah et al. (2007), Reijnders and Huijbregts (2008), and Menichetti and Otto (2008). The rationale for economic allocation is that demand is the driving force of production systems and thus their environmental burdens should be allocated according to market principles (Gnansounou et al., 2009). Compared to physical allocation, economic allocation produces results that are more rational when large quantities of by-products with low economic value are produced (Börjesson and Tufvesson, 2011). Nevertheless, the volatility of market prices, subsidies and market interferences are pointed out as the main drawbacks of this method, as they may strongly influence the calculation of allocation parameters and thus the results of the life-cycle study (Gnansounou et al., 2009). Finally, some authors (e.g. Huo et al., 2009) use a mix of allocation and/or substitution methods to address co-product credits in biofuel chains, i.e. they use a hybrid approach.

The issue of the most suitable allocation method is still open (Cherubini, 2010). In most studies no discussion is provided regarding the selection of the allocation procedure and, in general, no complete justification can be found concerning the reason to choose one and not a different allocation procedure. In fact, it is important to recognize that there is no single allocation procedure deemed appropriate for all biofuel processes (Mortimer et al., 2003). Therefore, whenever several alternative allocation procedures seem applicable, a sensitivity analysis should be conducted (ISO 14044:2006).

Several authors demonstrate that the choice and justification of allocation procedures are major issues in biofuel life-cycle studies, as they can have a significant influence on the results (Malça and Freire, 2004, 2006, 2010; Cherubini et al., 2009; Gnansounou et al., 2009; van der Voet et al., 2010). Moreover, the large influence of methodological choices (including allocation methods) may override many other types of uncertainty, as pointed out by Björklund (2002). This opinion is shared by Morgan and Henrion (1990) and Krupnick et al. (2006), who state that in some models the differences between scenarios may overcome parameter uncertainty and variability. Nevertheless, uncertainty due to choices cannot be eliminated, but can be rather easily illustrated by identifying the relevant alternatives and performing sensitivity analysis.

Section 3 presents an application of the approach presented and discussed in this section. Energy renewability efficiency and GHG intensity of rapeseed oil have been calculated capturing parameter uncertainty and alternative co-product treatment approaches.

3. An application to vegetable oil fuel in Europe

3.1 Vegetable oil use

Pure vegetable oil, also known as pure plant oil or straight vegetable oil, is an alternative fuel for diesel engines in transportation and also stationary applications, namely for heating purposes and/or electricity generation. The use of vegetable oils in internal combustion engines dates back to the beginning of the XX century, when a compression ignition engine, first developed by Rudolf Diesel, worked on peanut oil at the 1900's World Exhibition in Paris (Knothe, 2001). Vegetable oils were used in diesel engines for only a few years, however, until manufacturers optimized the engine design for low-grade fractions of petroleum in the 1920´s (Luque et al., 2008). Oil shortages in the 1930´s and 1970´s promoted once more research into the use of vegetable oil for energy purposes, as well as during World War II when vegetable oils were used as emergency fuels. An interesting aspect in the historical development and promotion of vegetable oils is that environmental issues were set aside and no emission studies were conducted (Knothe, 2001).

Main applications of vegetable oils include motor vehicles, e.g. passenger cars and agricultural machinery equipped with compression ignition engines, and stationary applications, like power generation (with diesel engine or gas turbine generators) and boiler heating systems (Cocco, 2009; Chiaramonti and Tondi, 2003). Concerning vehicle applications, vegetable oils represented almost 10% of the European biofuel consumption for road transport in 2007. Germany was the leading country in using motor fuels based on pure vegetable oils, with a consumption of approximately 1 Mtonnes in 2006 and 750 ktonnes in 2007 (EurObserv'ER, 2008). Other European countries using pure vegetable oils for automotive purposes include the Netherlands and Ireland, with several projects benefiting from excise duty relief and promoting the use of straight vegetable oil among transport operators, particularly vehicle fleets (DCENR, 2007; DMFA, 2007).

Direct utilization of vegetable oil does not pose significant technical challenges, if the energy system is conveniently adapted to the characteristics of the fuel (Chiaramonti and Tondi, 2003). Several properties of vegetable oils differ significantly from petroleum diesel, namely kinematic viscosity, thus requiring adjustments on fossil fuel-based systems for reliable operation. The kinematic viscosity of vegetable oils is about one order of magnitude greater, which thickens the lubricating oil and causes poor atomization in the combustion chamber, leading to incomplete combustion and carbon deposits (Chiaramonti and Tondi, 2003). The high viscosity of vegetable oils also limits their use during wintertime. To overcome the difficulties associated with straight vegetable oil (SVO) use, the viscosity must be reduced, which is usually achieved by preheating the vegetable oil. Typical conversion kits include a heat exchanger in which waste heat from the engine is supplied to the oil. This heat source can be supplemented by an electric booster (Mondal et al., 2008). An additional (small) fuel tank for fossil diesel and a three-way valve to switch between the main SVO tank and the small fossil diesel tank complete the package.

The need for modified diesel engines and the lack of a fuel distribution system are the main barriers for the dissemination of vegetable oil-powered vehicles. Moreover, there is no consensus on the suitability of SVO use in diesel vehicles equipped with direct injection engines or electronic injection pumps (Sidibé et al., 2010; Misra and Murthy, 2010). Large stationary applications, on the other hand, seem more feasible for the use of vegetable oil as diesel fuel. Actually, diesel engines and boilers that are able to burn low grade (and high viscosity) fuel oil can be easily switched to vegetable oil, owing to fuel and plant operation similarities. Furthermore, vegetable oils may not require complex or costly upgrading processes when used in heating systems. The main barrier for SVO penetration in large scale applications is rather economic, as a result of the low market prices of the fuel oils traditionally used (Chiaramonti and Tondi, 2003).

Several factors determine which vegetable oils are of most interest for energy purposes, namely geographic region, climate and economics. The main feedstock in cold, temperate European regions is rapeseed (canola); in the United States and some countries in Latin America soybean is the most used raw material; and in tropical countries, palm is the preferred feedstock. This article focuses on the direct use of rapeseed oil (pure or blended) as an energy carrier. Table 1 lists European data on rapeseed and rapeseed oil production, including the top 4 producing countries.

	2009 rapeseed production (10³ tonne)	World ranking [a]	2009 area harvested (10³ ha)	2009 crop yield (tonne/ha)	2005-2009 avg. crop yield [b] (tonne/ha)	2009 rapeseed oil production (10³ tonne)
Germany	6306.7	4th	1471.2	4.29	3.80 (3.44–4.29)	3345.3
France	5584.1	5th	1480.8	3.77	3.32 (2.90–3.77)	1742.6
Poland	2496.8	7th	810.0	3.08	2.75 (2.64–3.08)	870.8
UK	1951.0	8th	580.6	3.36	3.25 (3.10–3.36)	779.0
Total (EU-27)	**21417.6**	–	**6015.9**	**2.92**	–	**8466.7**
Top-4 share (%)	**76.3**	–	**72.2**	–	–	**79.6**

[a] World rankings for 2008; [b] minimum and maximum rapeseed yields in brackets.

Table 1. Rapeseed and rapeseed oil production in the EU-27, including major producers. (FAOSTAT, 2011; EUROSTAT, 2011)

Vegetable oils are currently used as diesel fuel for automotive purposes, and in thermal and power plants for heat and electricity production. Even though technological challenges for the use of straight vegetable oils have been overcome, there are still several non-technical barriers, namely the need for systems adaptation to run on SVO and the lack of a fuel distribution network, which do not occur with fossil fuels. Moreover, higher vegetable oil costs in comparison to fossil fuels also halt the market penetration of stationary SVO applications, as shown by the lower prices of heavy fuel oil for industry (Tables 2 and 3). For automotive applications, however, fuel costs work as an incentive for the promotion of SVO, with SVO prices lower than automotive fossil diesel prices.

On the other hand, the use of SVO seems very promising in developing countries, where self energy production at minimal costs is of greatest importance. The development of vegetable oil production chains, combining simpler production technology with lower production costs – e.g. mechanical oilseed presses, hand- or fuel-driven – is an approach that may greatly contribute for the socio-economic welfare of populations in these countries.

Year	Rapeseed oil
2005	669.4
2006	793.6
2007	970.0
2008	1329.2
2009	858.7
2010	951.1

[a] Prices paid at the farm gate.

Table 2. Annual average prices (US$/tonne) of rapeseed oil [a] (FAOSTAT, 2011)

Country	Heavy fuel oil for industry	Automotive diesel fuel [(a)]
Germany	515.2	1594.1
France	542.0	1483.5
Poland	590.0	1332.9
United Kingdom	n/a	1785.9

n/a: data not available; [(a)] A density of 0.85 kg/liter for diesel fuel has been used.

Table 3. Retail prices of selected fuels (US$/tonne) for the 1st quarter of 2010 (IEA, 2010)

3.2 Life-cycle modeling and inventory incorporating uncertainty

3.2.1 RO life-cycle chain

The life-cycle stages of the RO chain include rapeseed cultivation, harvesting, transport and drying of the seeds, crushing and extraction of the oil, oil degumming and refining. These steps are illustrated in the flowchart of Fig. 1. A detailed description of the RO production system can be found, for example, in Mortimer and Elsayed (2006), Stephenson et al. (2008) and Malça and Freire (2009, 2010).

Rape (Brassica napus L.), also known as Rapeseed, Oilseed Rape or Canola, is a yellow-flowered member of the family Brassicaceae widely cultivated throughout the world for the production of vegetable oil for human food consumption, but increasingly used for energy. Different cultivation methods may be used, namely in terms of soil management and soil inputs, depending on the climate region, soil type, and established agricultural practices. The cultivation step includes soil preparation, fertilization, sowing, weed control, and harvesting. Seeds are separated from the rest of the plant during harvesting. The straw, consisting of stalks, pods and leaves, is usually ploughed back into the field (SenterNovem, 2005; JEC, 2007; UFOP, 2008; Börjesson and Tufvesson, 2010). Several studies point out the incorporation of straw in the soil as a farm management activity with several benefits, namely the return and cycling of nutrients, the building of soil organic matter and the prevention of soil erosion.

Following harvesting, oilseeds are cleaned and dried. The typical moisture content of oilseeds is reduced, as required by oil extraction facilities and to ensure stability in storage. Moreover, large scale oil extraction is usually preceded by grinding and cooking of the seeds, to facilitate the oil extraction process. Vegetable oil may be extracted from the seeds by physical and/or chemical extraction. Different types of mechanical extraction devices can be used, namely the screw press and the ram press (Tickell et al., 2003). The first uses a screw inside a metal housing; as the screw turns, the oil is squeezed out of the seeds. The ram press uses a piston-cylinder set to crush the oilseeds. After mechanical pressing, protein-rich cake is also produced and can be used in animal feed. The press cake has, however, high oil content and a further (chemical) extraction step is usually conducted to extract the remaining oil, in order to increase the overall vegetable oil yield. Chemical extraction uses a petroleum-derived solvent, usually hexane; this is the extraction method considered in this chapter.

When solvent extraction is used, the oil goes through a distillation process to recover the hexane, which is recycled back to the oil extraction process. The final step in the production of vegetable oils is oil refining, which includes degumming, neutralization and drying. Gums are precipitated by the addition of hot water and phosphoric (or equivalent) acid and separated out by centrifugal separation. Free fatty acids in the oil are converted to soap using an alkali solution of sodium hydroxide, which is subsequently removed by continuous centrifugation. Finally, the oil is vacuum dried to remove any traces of water.

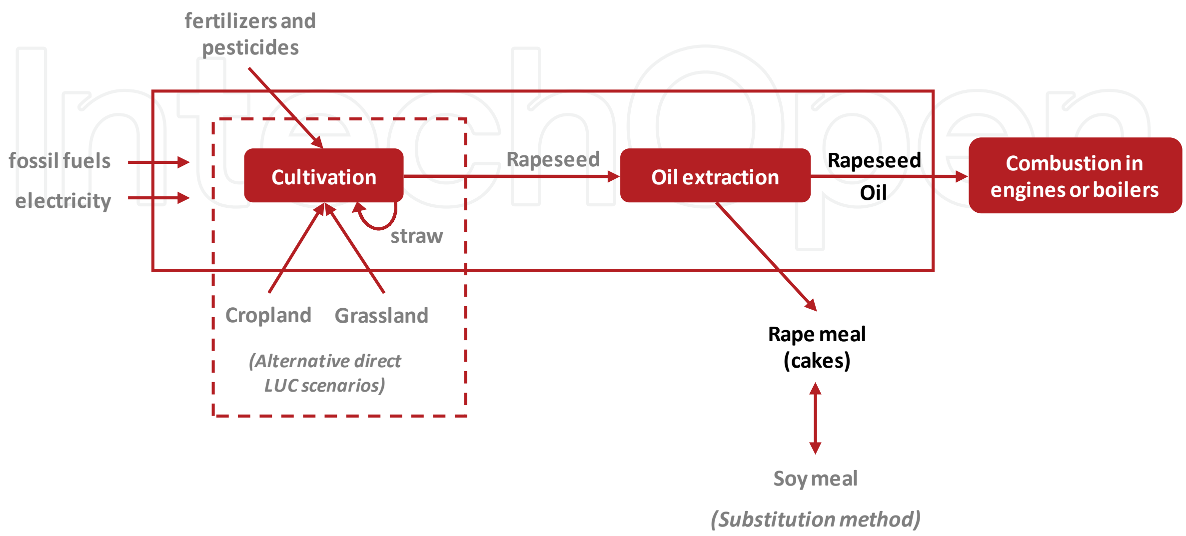


Fig. 1. Flow chart illustrating the life-cycle chain (well-to-tank) of Rapeseed Oil

The multifunctionality of biofuel systems is considered a critical issue in biofuel life-cycle studies, as discussed in section 2. For the RO production system, in particular, one valuable co-product is obtained: rape meal. Different approaches are addressed here for dealing with this co-production): i) the substitution method, in which the system is expanded with the avoided process – (soy meal production); ii) allocation, i.e. splitting up the process into two single-functional processes (RO production + rape meal production) on the basis of underlying relationships (physical: mass, energy; and economic); and iii) the no allocation, in which rape meal is ignored, i.e. all burdens (energy and material inputs, and related emissions) are fully allocated to RO.

Concerning the application of the substitution method, it is considered that the RO co-product rapeseed meal replaces imported soybean meal in animal feed. The technical feasibility of replacing soybean meal with rapeseed meal for feeding pigs and piglets has already been demonstrated (e.g. Kracht et al., 2004). Research recently conducted in France has also concluded that replacing soybean meal with rapeseed meal in the feed rations for dairy cows and for fattening beef cattle is technically feasible (GAIN, 2005). Actually, rape meal from oilseed crushing is replacing soybean meal imports as a high-protein animal feed (GAIN, 2007; Ceddia and Cerezo, 2008). This substitution approach is also considered in other works (e.g. Bernesson et al., 2004; JEC, 2007; Lechón et al., 2009; Soimakallio et al., 2009).

3.2.2 Key issues affecting soil carbon exchange

Several issues influence soil carbon exchange, namely land use change scenarios, agricultural practices and geographic region. Concerning land use change, two reference land uses have been considered in this article: (i) grassland; and (ii) long-term cultivated cropland. Appropriate land use factors F_{LU}, which reflect the difference in soil organic

carbon associated with the type of land use compared to a standard soil organic carbon SOC_{ST}, have been taken from EC (2010), IPCC (2006).
EC (2010) differentiates three alternative management practices for cropland – full-tillage; reduced or low-tillage; and no-till – based on the level of soil disturbance during cultivation, respectively substantial, reduced or minimal. Full- and reduced-tillage have been considered for the reference land use, whereas low-tillage has been assumed for the actual land use (rapeseed cultivation). Concerning grassland, the management scenario that most contributes to carbon sequestration in the soil is improved grassland (according to EC 2010), which has been used in our assessment. The alternatives in soil management practices have been quantified through F_{MG}, a factor that reflects the difference between the soil organic carbon associated with the main management practice and the standard soil organic carbon SOC_{ST} (EC, 2010a).
The level of carbon input to the soil may also differ depending on the return of crop residues to the field and the adoption of other agricultural practices (EC, 2010a). To quantify extreme scenarios in terms of soil carbon content in the reference land use, high and low carbon inputs have been considered, respectively for grassland and cropland, whereas in the actual land use the option for medium inputs to rapeseed cultivation has been selected. The input factor F_I, which reflects the difference in soil organic carbon associated with different levels of carbon input to soil compared to the standard soil organic carbon SOC_{ST}, has been used (EC, 2010a; IPCC, 2006).
The geographic region is another key aspect for assessing the GHG emissions of a specific crop, since climate and soil type are two important factors affecting the calculation of land carbon stocks. Main rapeseed oil producers in Europe are France and Germany (see Table 1). A cool temperate moist climate has been selected as representative of main rapeseed production in Europe, according to the classification made in EC (2010). Concerning soil type, EC (2010) shows that high activity clay soil is the most representative soil type for countries involved in rapeseed cultivation. Active soils are also indicated in JEC (2007) as the most likely soil type to be converted to arable cropping.

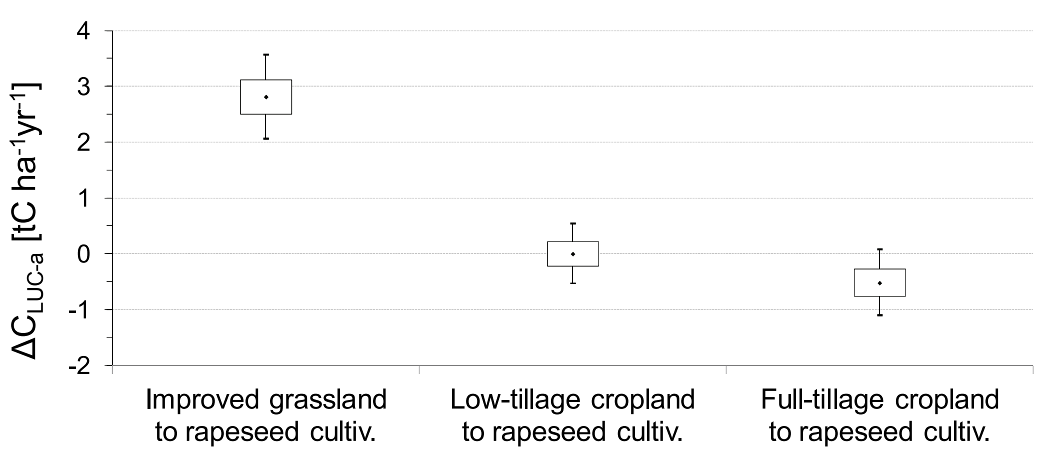


Fig. 2. Soil carbon exchange associated with LUC scenarios for Rapeseed Oil. The boxes show the interquartile range, the mark is the median and the ends of the whiskers are the 5th and 95th percentiles. Same notation is used in figs. 3 and 5

As shown in the above discussion, a large degree of variability exists concerning the management practices and input levels associated with rapeseed cultivation. The guidance provided in EC (2010) concerning the selection of the appropriate coefficients F_{LU}, F_{MG} and F_I for land use and management has been followed in this article. Moreover, appropriate probability distributions have been assigned to ΔC_{LUC-a}, based on the error ranges provided in IPCC (2006) for each LUC scenario (Fig. 2).

3.3 Results and discussion

Rapeseed oil life-cycle energy renewability efficiency and GHG intensity incorporating uncertainty are presented in section 3.3. GHG emission savings of displacing petroleum diesel are also evaluated. As discussed in section 2, a "well-to-tank" approach has been used, in which energy and GHG emissions are assessed from the very first production stage until the final fuel distribution depot. The functional unit chosen is 1 MJ of fuel energy content (FEC), measured in terms of the lower heating value (LHV).

3.3.1 Energy Renewability Efficiency

The life-cycle energy renewability efficiency ERenEf of rapeseed oil is displayed in the box plot of Fig. 3. The output distributions are divided in the 5^{th}, 25^{th}, 50^{th}, 75^{th}, and 95^{th} percentiles. Scenario uncertainty has been considered regarding the modeling choice of how co-product credits are accounted for, namely using mass, energy and market value allocation approaches and the substitution method. A comparison with fossil diesel shows that rapeseed oil clearly contributes to non-renewable primary energy savings as opposed to its fossil reference. RO ERenEf is clearly positive, which indicates that an important fraction of the biofuel energy content (from 60% to 85%, depending on the approach for dealing with co-products, Fig. 3) comes from renewable energy sources.
Comparing the three allocation methods used, Fig. 3 shows that mass allocation results have the lowest uncertainty range, whereas economic allocation results are more uncertain because they depend on the variability of market prices. System expansion shows the highest degree of uncertainty due to differences in credits for soy meal substitution by rape meal.

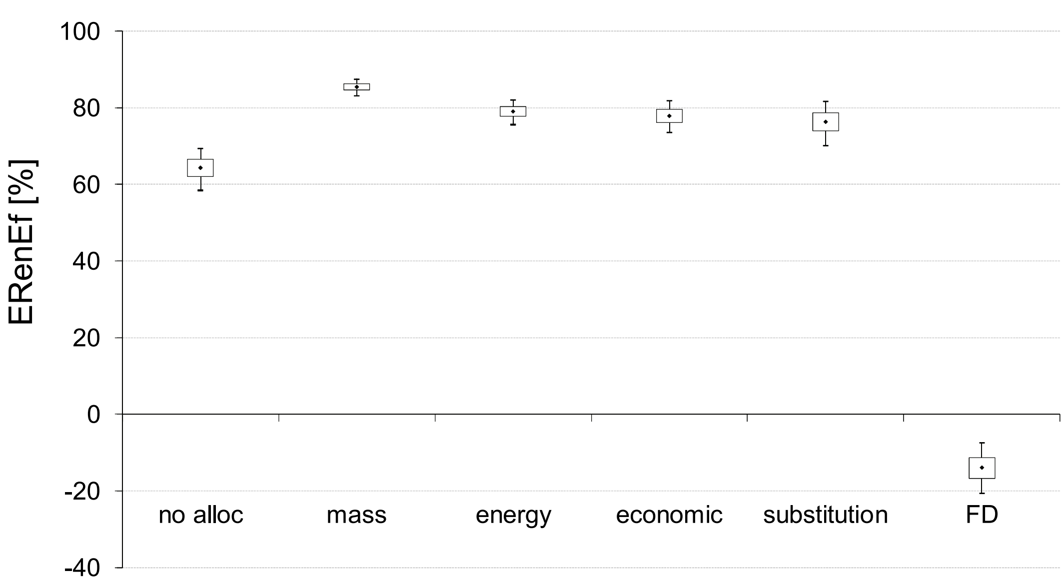
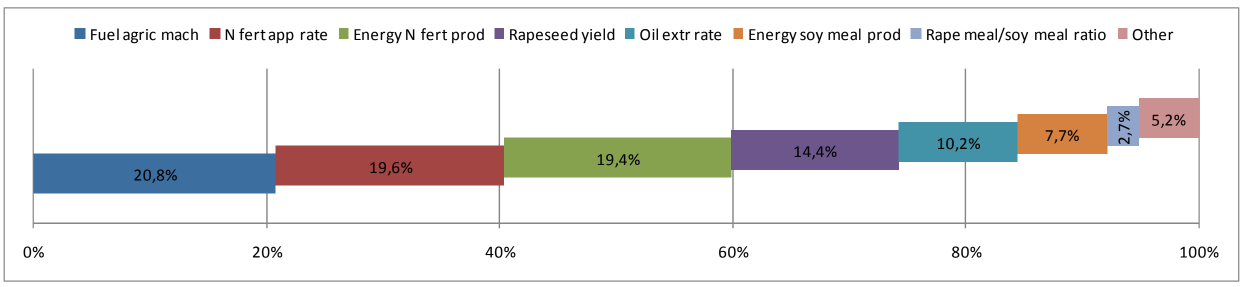
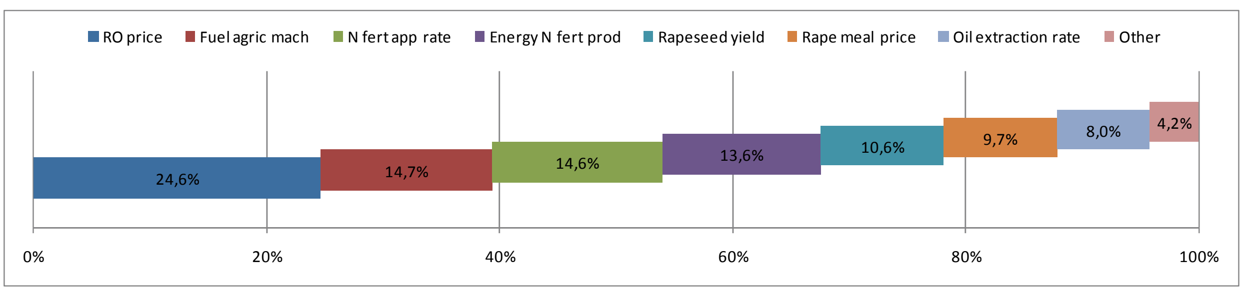


Fig. 3. RO life-cycle ERenEf results: scenario and parameter uncertainty

Moreover, mass allocation shows the highest results, which is explained by the relatively high mass share of rape meal in the oil extraction stage (approximately 1.5 kg of rape meal per kg of RO produced). Although it is a straightforward method, mass allocation is very often a meaningless approach, namely when energy systems or market principles come into play. Allocations based on energy and economic value show lower ERenEf values, due to the higher heating value and market price of RO in comparison to rape meal.
Figure 4 shows which parameters are most significant in the overall uncertainty of RO ERenEf. The uncertainty importance analysis that has been conducted shows that several parameters have important contributions in the uncertainty, namely diesel fuel use in agricultural machinery, N fertilizer application rate and energy use in N fertilizer production. In particular, Fig. 4(b) for economic allocation shows that market prices (and their inherent volatility) also affect the variance of ERenEf.



(a)



(b)

Fig. 4. Contribution of input data to the variance of RO life-cycle ERenEf: (a) substitution method; (b) economic allocation

3.3.2 GHG savings

Life-cycle GHG emission savings of RO displacing petroleum diesel are shown in Fig. 5. The uncertainty associated with the life-cycle GHG emissions of petroleum diesel has been considered using a normal probability distribution (μ=82 g CO_2eq MJ^{-1}; σ=3 g CO_2eq MJ^{-1}). An important conclusion from Fig. 5 is that parameter uncertainty is significantly higher in the case of RO GHG emissions when compared to ERenEf values of Fig. 3. An uncertainty importance analysis will put into evidence the parameters that most contribute to this higher magnitude of uncertainty.
Figure 5 shows that RO GHG emissions are considerably higher than fossil diesel (FD) GHG emissions if the most severe land use change scenario (improved grassland to rapeseed cultivation) is considered, i.e. FD substitution by RO results in negative GHG savings. This outcome contrasts with the other two LUC scenarios (conversion from full-tillage or low-

tillage croplands) in which rapeseed oil GHG savings are positive. Moreover, these savings are above the 35% GHG saving target of the European renewable energy directive (EPC, 2009), regardless of the co-product method used.
Fig. 5 also shows that in the "low-tillage cropland to rapeseed cultivation" LUC scenario, the parameter uncertainty range overcomes the differences between calculated median values for the various scenarios of co-product treatment. Soil carbon sequestration associated with conversion of "full-tillage cropland to rapeseed cultivation" results in very low RO life-cycle GHG emissions, complying with the 2018 target of 60% GHG savings over fossil diesel of EPC (2009). In this scenario, differences between co-product approaches become negligible.

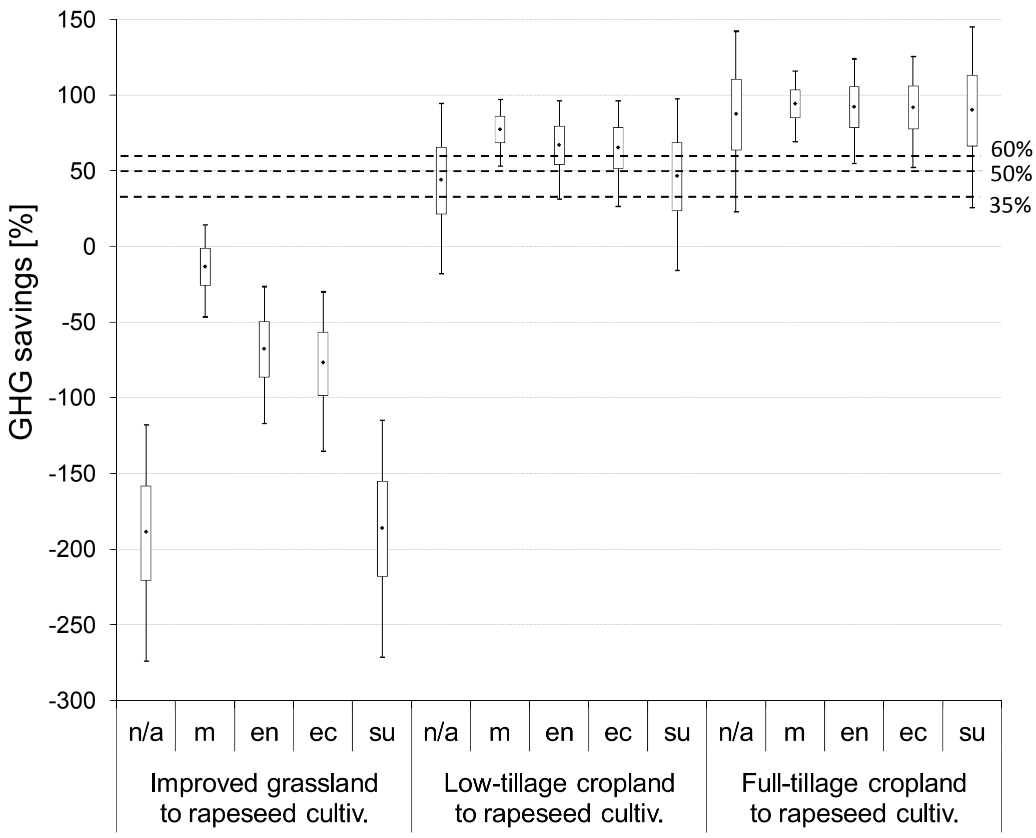
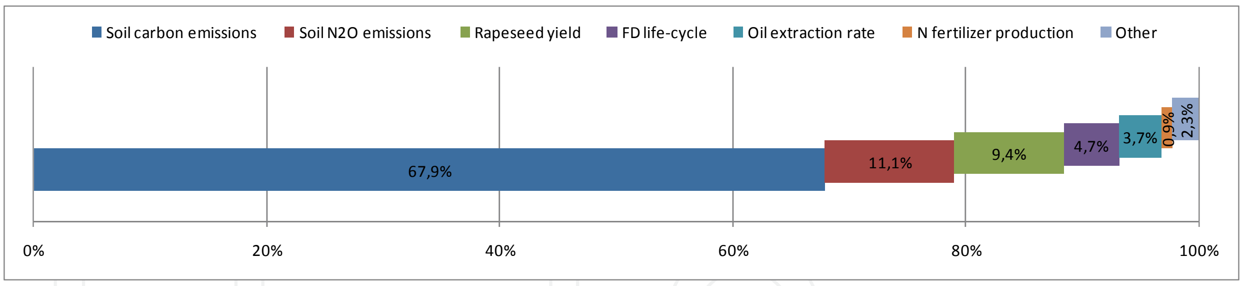
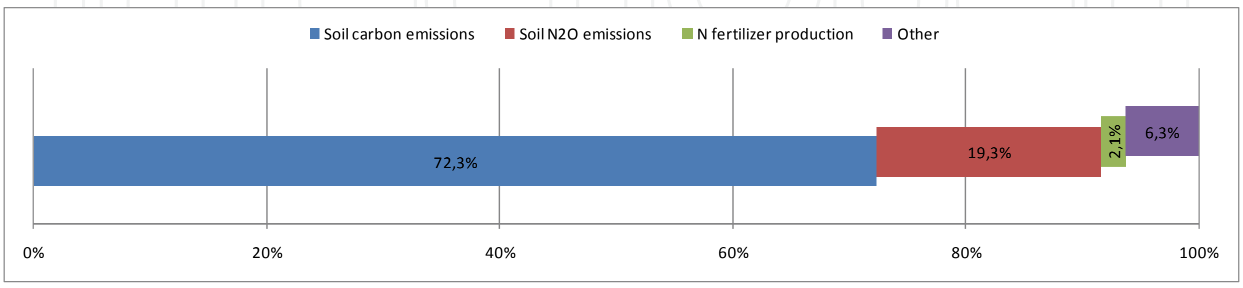


Fig. 5. RO life-cycle GHG emission savings: LUC scenarios and co-product approaches (n/a: no allocation; m: mass; en: energy; ec: economic; su: substitution). Dashed lines indicate minimum levels of GHG savings (EPC, 2009)

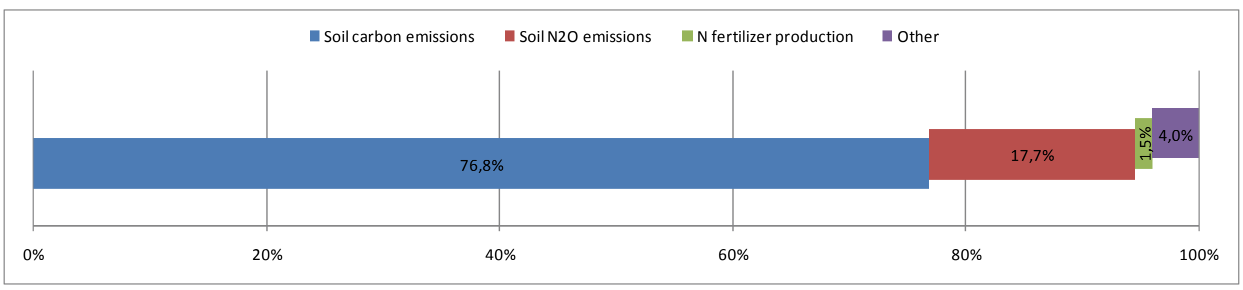
Figure 6 shows which parameters are most significant in the overall uncertainty of RO GHG emissions for the three LUC scenarios considered. The highest sources of uncertainty arise in the cultivation stage. Soil carbon emissions from land use change are the main contributor to the uncertainty of RO GHG intensity, with nitrous oxide emissions from cultivated soil as the second most important aspect. Agricultural yield and oil extraction efficiency (amount of rapeseed oil that can be extracted per kg of processed seed) are also important in the "grassland to rapeseed" LUC scenario. The remaining parameters hardly contribute to the variance of GHG emissions. Further research work must focus on the most important sources of uncertainty, in order to reduce the overall uncertainty of the rapeseed oil chain and improve the reliability of RO life-cycle studies outcomes.



(a)



(b)



(c)

Fig. 6. Contribution of input data to the variance of RO life-cycle GHG emission savings (substitution method). Land use change scenarios: (a) improved grassland to rapeseed cultivation; (b) low-tillage cropland to rapeseed cultivation; (c) full-tillage cropland to rapeseed cultivation

4. Conclusions

This chapter has two main goals: i) to present a robust framework to incorporate uncertainty in the life-cycle modeling of biofuel systems; and ii) to describe the application of the framework to vegetable oil fuel in Europe. The chapter also compares rapeseed oil life-cycle results (energy renewability efficiency and GHG emissions) with its fossil fuel equivalent (diesel), in order to evaluate potential savings achieved through displacement.

A comprehensive assessment of uncertainty in the life-cycle of rapeseed oil has been conducted. Several sources of uncertainty have been investigated, namely related to parameters, global warming potentials and concerning how co-product credits are accounted for. It has been shown that depending on whether or not uncertainty in parameters is taken into account, and what modeling choices are made, results and conclusions from the life-cycle study may vary quite widely. In particular, it has been reported that the net GHG balance is strongly influenced by soil carbon stock variations due to land use change and by the magnitude of nitrous oxide emissions from cultivated soil.

Depending on prior land use, GHG emissions may comply with the European directive target of 35% GHG emission savings or, conversely, may completely offset carbon gains attributed to rapeseed oil production. These results contrast with the energy balance of rapeseed oil, which shows a high degree of energy renewability efficiency, regardless of parameter uncertainty and modeling choices made. Moreover, non-renewable primary energy savings are always achieved with rapeseed oil use, as opposed to fossil diesel use.
The benefits of using rapeseed oil to displace fossil diesel have been demonstrated, but special attention is needed to reduce emissions from carbon stock changes and nitrogen fertilizer application, in order to ensure that rapeseed oil use avoids GHG emissions. Only through a comprehensive evaluation of the life-cycle of biofuels, capturing uncertainty issues, it is possible to ensure reliable outcomes and guarantee the environmental sustainability of biofuel production systems.

5. Acknowledgements

The research presented in this article has been supported by the Portuguese Science and Technology Foundation (FCT) projects PTDC/TRA/72996/2006 "*Biofuel systems for transportation in Portugal: a well-to-wheels integrated multi-objective assessment*", MIT/SET/0014/2009 "*Biofuel capturing uncertainty in biofuels for transportation: resolving environmental performance and enabling improved use*", and MIT/MCA/0066/2009 "*Economic and Environmental Sustainability of Electric Vehicle Systems*".

6. References

ADEME (Agence de l'Environnement et de la Maitrise de l'Energie). Energy and greenhouse gas balances of biofuels' production chains in France, executive summary, Paris; December 2002.

Anex R, Lifset R. 2009. Assessing Corn Ethanol: Relevance and Responsibility. *Journal of Industrial Ecology* 13(4):479-482.

Armstrong A, Baro J, Dartoy J, Groves A, Nikkonen J, Rickeard D, Thompson D, & Larivé J. Energy and greenhouse gas balance of biofuels for Europe - an update, report no. 2/02. Brussels: CONCAWE, 2002.

Bernesson S, Nilsson D, & Hansson PA. 2004. A limited LCA comparing large- and small-scale production of rape methyl ester (RME) under Swedish conditions. *Biomass & Bioenergy* 26(6):545–559.

Björklund A. 2002. Survey of Approaches to Improve Reliability in LCA. *Int. Journal of Life Cycle Assessment* 7(2):64-72.

Börjesson P, & Tufvesson L. 2011. Agricultural crop-based biofuels – resource efficiency and environmental performance including direct land use changes. *Journal of Cleaner Production* 19:108-120.

Boustead I, & Hancock G. Handbook of Industrial Energy Analysis. Ellis Horwood ltd, John Wiley and Sons, 1979.

Boustead I. Eco-Profiles of the European plastics industry. Methodology. Report. Brussels: Association of Plastics Manufacturers in Europe, 2003.

Bowyer C. 2010. Anticipated Indirect Land Use Change Associated with Expanded Use of Biofuels and Bioliquids in the EU – An Analysis of the National Renewable Energy Action Plans. Institute for European Environmental Policy, London, UK.

Ceddia M, & Cerezo E. A Descriptive Analysis of Conventional Organic and GM crop and Certified Seed Production in the EU. Luxembourg: Joint Research Centre of the European Commission; 2008.

Cherubini F, Birda N, Cowie A, Jungmeier G, Schlamadinger B, & Woess-Gallasch S. 2009. Energy- and greenhouse gas- based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. *Resources Conservation & Recycling* 53(8):434-447.

Cherubini F. 2010. GHG balances of bioenergy systems – Overview of key steps in the production chain and methodological concerns. Renewable Energy 35:1565–1573.

Cherubini F, & Strømman AH. 2011. Life cycle assessment of bioenergy systems: State of the art and future challenges. *Bioresource Technology* 102:437-451.

Chiaramonti D, & Tondi G. Stationary Applications of Liquid Biofuels, Final Report, ETA Renewable Energies, December, Firenze, 2003.

Ciroth A, Fleischer G, & Steinbach J. 2004. Uncertainty Calculation in Life Cycle Assessments: A Combined Model of Simulation and Approximation. *International Journal of Life Cycle Assessment* 9(4): 216-226.

Cocco D. 2009. Predicted performance of integrated power plants based on diesel engines and steam cycles fuelled with a rapeseed oil chain. Proceedings of the Institution of Mechanical Engineers Part A – *Journal of Power and Energy* 223(A5):477–485.

Croezen H, Bergsma G., Otten M., & van Valkengoed M. Biofuels: indirect land use change and climate impact. CE Delft, Delft, the Netherlands, June 2010.

Crutzen PJ, Mosier AR, Smith KA, & Winiwarter W. 2008. N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmospheric Chemistry and Physics* 8(2):389–395.

DCENR (Department of Communications, Energy and Natural Resources). Report on measures taken to promote the use of biofuels or other renewable fuels to replace diesel or petrol: compliance with Directive 2003/30/EC. July, Ireland, 2007.

Dewulf A, van Langenhove H, & van de Velde B. 2005. Exergy-Based Efficiency and Renewability Assessment of Biofuel Production. *Environmental Science & Technology* 39:3878-3882.

DMFA (Dutch Ministry of Foreign Affairs). Report from the Netherlands for 2006 pursuant to Article 4(1) of Directive 2003/30/EC on the promotion of the use of biofuels or other renewable fuels for transport. July, The Netherlands, 2007.

EC (European Commission). 2010a. Commission decision 2010/335/EU of 10 June 2010 on guidelines for the calculation of land carbon stocks for the purpose of Annex V to Directive 2009/28/EC.

EC 2010b. Report from the Commission on indirect land-use change related to biofuels and bioliquids. COM(2010) 811 final, December 22.

EPA (Environmental Protection Agency). Regulation of Fuels and Fuel Additives: Modifications to Renewable Fuel Standard Program. Federal Register, Vol. 75, No. 244, December 21, 2010.

EPC. Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC; 2009.

EurObserv'ER. Biofuels Barometer. Systèmes Solaires - Le journal des énergies renouvelables 185; June 2008, p. 49-66.

Eurostat (Statistical Office of the European Communities). http://epp.eurostat.ec.europa.eu (accessed March 2011).

FAOSTAT (FAO statistical database). http://faostat.fao.org (accessed January 2011).

Fargione J, Hill J, Tilman D, Polasky S, & Hawthorne P. 2008. Land Clearing and the Biofuel Carbon Debt. *Science* 319(5867):1235-1238.

Farrell AE, Plevin RJ, Turner BT, Jones AD, O'Hare M, & Kammen DM. 2006. Ethanol Can Contribute to Energy and Environmental Goals. *Science* 311, pp. 506-508, Jan 27.

Frischknecht R. 2000. Allocation in Life Cycle Inventory Analysis for Joint Production. *Int. Journal of Life Cycle Assessment* 5 (2):85-95.

Fritsche U, Sims R, & Monti A. 2010. Direct and indirect land-use competition issues for energy crops and their sustainable production – an overview. *Biofuels, Bioproducts & Biorefining* 4:692–704.

GAIN (Global Agriculture Information Network). 2005. France explores substituting soybean meal with rapeseed meal. USDA Foreign Agricultural Service. Paris.

GAIN (Global Agriculture Information Network). Impacts on Oilseed Industry following Biofuel Boom. Paris: USDA Foreign Agricultural Service; 2007.

Geisler G, Hellweg S, & Hungerbuhler K. 2005. Uncertainty Analysis in Life Cycle Assessment (LCA): Case Study on Plant-Protection Products and Implications for Decision Making. *International Journal of Life Cycle Assessment* 10(3): 184-192.

Gnansounou E, Dauriat A, Villegas J, & Panichelli L. 2009. Life cycle assessment of biofuels: Energy and greenhouse gas balances. *Bioresource Technology* 100(21):4919-4930.

Guinée JB, Heijungs R, & Huppes G. 2004. Economic Allocation: Examples and Derived Decision Tree. *Int Journal of Life Cycle Assessment* 9(1):23-33.

Guinée JB, Heijungs R, & van der Voet E. 2009. A greenhouse gas indicator for bioenergy: some theoretical issues with practical implications. *Int Journal of Life Cycle Assessment* 14(4):328-339.

Heijungs R, & Huijbregts M. 2004. A Review of Approaches to Treat Uncertainty in LCA. In: C. Pahl-Wostl, S. Schmidt, A.E. Rizzoli, and A.J. Jakeman (Eds). Complexity and Integrated Resources Management. Transactions of the 2nd Biennial Meeting of the International Environmental Modelling and Software Society, Vol 1, Osnabrück.

Heijungs, R. 1996. Identification of key issues for further investigation in improving the reliability of life-cycle assessments. *Journal of Cleaner Production* 4(3–4):159–166.

Hekkert M, Hendriks F, Faaij A, & Neelis M. 2005. Natural gas as an alternative to crude oil in automotive fuel chains well-to-wheel analysis and transition strategy development. *Energy Policy* 33:579–594.

Heller M, Keoleian G, & Volk T. 2003. Life cycle assessment of a willow bioenergy cropping system. *Biomass and Bioenergy* 25:147–165.

Hoefnagels R, Smeets E, & Faaij A. 2010. Greenhouse gas footprints of different biofuel production systems. *Renewable & Sustainable Energy Reviews* 14:1661-1694.

Huijbregts, M. 1998. Application of uncertainty and variability in LCA. Part I: A General Framework for the Analysis of Uncertainty and Variability in Life Cycle Assessment. *International Journal of Life Cycle Assessment* 3(5): 273–280.

Huijbregts M, Gilijamse W, Ragas A, & Reijnders L. 2003. Evaluating Uncertainty in Environmental Life-Cycle Assessment. A Case Study Comparing Two Insulation Options for a Dutch One-Family Dwelling. *Environmental Science & Technology* 37:2600-2608.

Huijbregts M, Rombouts L, Hellweg S, Frischknecht R, Hendriks A, van de Meent D, Ragas A, Reijnders L, & Struijs J. 2006. Is Cumulative Fossil Energy Demand a Useful Indicator for the Environmental Performance of Products? *Environmental Science & Technology* 40(3):641-648.

Huo H, Wang M, Bloyd C, & Putsche V. 2009. Life-Cycle Assessment of Energy Use and Greenhouse Gas Emissions of Soybean-Derived Biodiesel and Renewable Fuels. *Environmental Science & Technology* 43:750-756.

IEA (International Energy Agency). 2010 key world energy statistics, Paris, 2010.

IPCC (Intergovernmental Panel on Climate Change). Climate Change 2007 - The Physical Science Basis. Fourth Assessment Report. New York: Cambridge University Press; 2007.

IPCC. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Vol.4, Ch. 11: N2O Emissions from Managed Soils and CO_2 Emissions from Lime and Urea Application. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K, editors. Japan: Institute for Global Environmental Strategies; 2006.

ISO (International Organization for Standardization). ISO 14040: Environmental management – Life cycle assessment – Principles and framework. Genève, Switzerland; 2006.

ISO. ISO 14044: Environmental management – Life cycle assessment – Requirements and guidelines. Genève, Switzerland; 2006.

Janulis P. 2004. Reduction of energy consumption in biodiesel fuel life cycle. *Renewable Energy* 29:861-871.

JEC (JRC/EUCAR/CONCAWE Consortium; CONCAWE: The oil companies' European association for environment, health and safety in refining and distribution; EUCAR: European Council for Automotive R&D; JRC: Joint Research Centre of the European Commission). Well-to-wheels analysis of future automotive fuels and powertrains in the European context. Well-to-tank and Well-to-wheels report, Version 2c, Brussels; March 2007.

Kaiser E, Kohrs K, Kucke M, Schnug E, Heinemeyer O, & Munch J. 1998. Nitrous oxide release from arable soil: importance of N-fertilization, crops and temporal variation. *Soil Biology & Biochemistry* 30(12):1553-1563.

Kim S, & Dale B. 2002. Allocation Procedure in Ethanol Production System from Corn Grain. I-System Expansion. *Int. Journal of Life Cycle Assessment* 7(4):237-243.

Kløverpris J, Wenzel H, Banse M, Milà i Canals L, & Reenberg A. 2008. Conference and Workshop on Modelling Global Land Use Implications in the Environmental Assessment of Biofuels. *Int Journal of Life Cycle Assessment* 13(3):178-183.

Knothe G. Historical perspectives on vegetable oil-based diesel fuels. *Industrial Oils* 12: 1103-1107, 2001.

Kracht W, Nicke S, Kluge H, Keller K, Matzke W, Hennig U & Schumann W. 2004. Effect of Dehulling of Rapeseed on Feed Value and Nutrient Digestibility of Rape Products in Pigs. *Archives of Animal Nutrition* 58(5):389-404.

Krupnick A, Morgenstern R, Batz M, Nelson P, Burtraw D, Shih J, & McWilliams M. 2006. Not a sure thing: Making regulatory choices under uncertainty. Technical report, Resources for the Future, Washington DC.

Larson E. 2006. A review of life-cycle analysis studies on liquid biofuel systems for the transport sector. *Energy & Sustainable Development* 10(2):109–126.

Lechón Y, Cabal H, de la Rúa C, Caldés N, Santamaría M, & Sáez R. 2009. Energy and greenhouse gas emission savings of biofuels in Spain's transport fuel. The adoption of the EU policy on biofuels. *Biomass & Bioenergy* 33(6-7):920-932.

Liska AJ, & Cassman KG. 2008. Towards Standardization of Life-Cycle Metrics for Biofuels: Greenhouse Gas Emissions Mitigation and Net Energy Yield. *Journal of Biobased Materials and Bioenergy* 2:187–203.

Liska AJ, & Perrin RK. 2009. Indirect land use emissions in the life cycle of biofuels: regulations vs science. *Biofuels, Bioproducts & Biorefining* 3:318–328.

Lloyd S, & Ries R. 2007. Characterizing, Propagating, and Analyzing Uncertainty in Life-Cycle Assessment: A Survey of Quantitative Approaches, *Journal of Industrial Ecology* 11(1): 161–179.

Luque R, Davila L, Campelo JM, Clark JH, Hidalgo JM, Luna D et al. 2008. Biofuels: a technological perspective. *Energy and Environmental Science* 1(5):542-564.

Luo L, van der Voet E, Huppes G, & Udo de Haes H. 2009 Allocation issues in LCA methodology: a case study of corn stover-based fuel ethanol. *Intl Journal of Life Cycle Assessment* 14:529-539.

Malça J, & Freire F. 2004. Life cycle energy analysis for bioethanol: allocation methods and implications for energy efficiency and renewability. 17th International Conference on Efficiency, Costs, Optimization, Simulation and Environmental Impact of Energy and Process Systems (ECOS 2004), July 07-09, Guanajuato Mexico.

Malça J, & Freire F. 2006. Renewability and life-cycle energy efficiency of bioethanol and bioethyl tertiary butyl ether (bioETBE): Assessing the implications of allocation. *Energy* 31(15):3362-3380.

Malça J, & Freire F. 2009. Energy and environmental benefits of rapeseed oil replacing diesel. *International Journal of Green Energy* 6(3):287-301.

Malça J, & Freire F. 2010. Uncertainty Analysis in Biofuel Systems: An Application to the Life Cycle of Rapeseed Oil. *Journal of Industrial Ecology* 14(2):322-334.

Malça J, & Freire F. 2011. Life-cycle studies of biodiesel in Europe: A review addressing the variability of results and modeling issues. *Renewable & Sustainable Energy Reviews* 15(1):338-351.

Menichetti E, & Otto M. 2008. Energy Balance & Greenhouse Gas Emissions of Biofuels from a Life Cycle Perspective. In: R.W. Howarth and S. Bringezu (eds) Biofuels: Environmental Consequences and Interactions with Changing Land Use. Proceedings of the Scientific Committee on Problems of the Environment (SCOPE) International Biofuels Project Rapid Assessment, September 22-25 2008, Gummersbach, Germany.

Misra R, & Murthy M. 2010. Straight vegetable oils usage in a compression ignition engine—A review. *Renewable and Sustainable Energy Reviews* 14:3005–3013.

Mondal P, Basu M, & Balasubramanian N. 2008. Direct use of vegetable oil and animal fat as alternative fuel in internal combustion engine. *Biofuels, Bioproducts & Biorefining* 2:155–174.

Morgan MG & Henrion M. 1990. A Guide to Dealing with Uncertainty in Quantitative Risk and Policy Analysis. New York: Cambridge University Press.

Mortimer N, Cormack P, Elsayed M, & Horne R. Evaluation of the Comparative Energy, Global Warming and Social Costs and Benefits of Biodiesel, report. UK: Resource Research Unit, Sheffield Hallam University, 2003.

Mortimer ND, Elsayed MA. North East Biofuel Supply Chain Carbon Intensity Assessment. Sheffield, UK: North Energy Associates Ltd; 2006.

Neupane B, Halog A, & Dhungel S. 2011. Attributional life cycle assessment of woodchips for bioethanol production. *Journal of Cleaner Production* 19:733-741.

Oracle. 2010. Oracle Crystal Ball software v.11.1.

Papong S, & Malakul P. 2010. Life-cycle energy and environmental analysis of bioethanol production from cassava in Thailand. *Bioresource Technology* 101:S112–S118.

Plevin R. 2010. Life Cycle Regulation of Transportation Fuels: Uncertainty and its Policy Implications. PhD thesis. University of California, Berkeley, USA.

Poldy F. 2008. Net energy and strategic decision-making. *Biofuels, Bioproducts & Biorefining* 2:389–392.

Rabl A, Benoist A, Dron D, Peuportier B, Spadaro J, & Zoughaib A. 2007. How to Account for CO_2 Emissions from Biomass in an LCA. *Intl Journal of Life Cycle Assessment* 12(5):281.

Reijnders L, & Huijbregts M. 2008. Biogenic greenhouse gas emissions linked to the life cycles of biodiesel derived from European rapeseed and Brazilian soybeans. *Journal of Cleaner Production* 16(18):1943–1948.

Reijnders L. 2009. Transport biofuels: Can they help limiting climate change without an upward impact on food prices? *Journal of Consumer Protection and Food Safety* 4:75-78.

Saltelli A, Ratto M, Tarantola S, & Campolongo F. 2006. Sensitivity analysis practices: Strategies for model-based inference. *Reliability Engineering & System Safety* 91(10-11):1109–1125.

Schade B, & Wiesenthal T. 2011. Biofuels: A model based assessment under uncertainty applying the Monte Carlo method. *Journal of Policy Modeling* 33:92–126.

Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, et al. 2008. Use of U.S. Croplands for Biofuels Increases Greenhouse Gases through Emissions from Land-Use Change. *Science* 319(5867):1238-1240.

SenterNovem (Agency of the Dutch Ministry of Economic Affairs for Innovation and Sustainable Development). The road to pure plant oil? The technical, environment-hygienic and cost-related aspects of pure plant oil as a transport fuel. Report 2GAVE-05.05. The Netherlands, 2005.

Shapouri H, Duffield J, & Graboski M. Estimating the Net Energy Balance of Corn Ethanol, report no. 721. US Dept. of Agriculture, 1995.

Shapouri H, Duffield J, & Wang M. The Net Energy Balance of Corn Ethanol: an Update, report no. 813. US Dept. of Agriculture, 2002.

Sheehan J, Camobreco V, Duffield J, Graboski M, & Shapouri H. Life Cycle Inventory of Biodiesel and Petroleum Diesel for Use in an Urban Bus, Final Report. Golden, CO: National Renewable Energy Laboratory, 1998.

Sidibé S, Blin J, Vaitilingom G, & Azoumah Y. 2010. Use of crude filtered vegetable oil as a fuel in diesel engines state of the art: Literature review. *Renewable and Sustainable Energy Reviews* 14:2748–2759.

Soimakallio S, Makinen T, Ekholma T, Pahkala K, Mikkola H, & Paappanen T. 2009. Greenhouse gas balances of transportation biofuels, electricity and heat generation in Finland: Dealing with the uncertainties. *Energy Policy* 37:80-90.

Stephenson AL, Dennis JS, & Scott SA. 2008. Improving the sustainability of the production of biodiesel from oilseed rape in the UK. *Process Safety and Environmental Protection* 86:427-440.

Tickell J. 2003. From the fryer to the fuel tank – The complete guide to using vegetable oil as an alternative fuel, 3rd ed., Joshua Tickell Publications, New Orleans, Louisiana.

UFOP (Union zur Förderung von Oel-und Proteinpflanzen e.V.). *Rapeseed Magazine*, Rape Blossom, 2008.

van der Voet E, Lifset RJ, & Luo L. 2010. Life-cycle assessment of biofuels, convergence and divergence. *Biofuels* 1(3):435-449.

Wagner U, Eckl R, & Tzscheutschler P. 2006. Energetic life cycle assessment of fuel cell powertrain systems and alternative fuels in Germany. *Energy* 31(14):3062-3075.

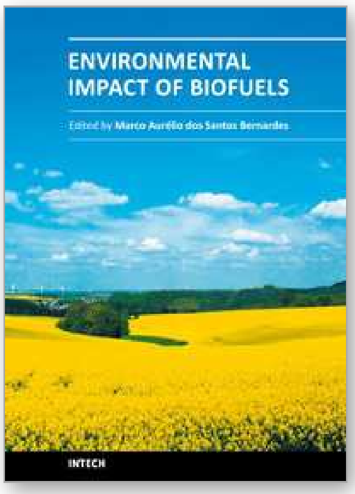
Weidema B, Fress N, Petersen E, & Ølgaard H. Reducing Uncertainty in LCI: Developing a Data Collection Strategy, Environmental Project No. 862. Denmark, 2003.

Whitaker J, Ludley KE, Rowe R, Taylor G, & Howard DC. 2010. Sources of variability in greenhouse gas and energy balances for biofuel production: a systematic review. *Global Change Biology Bioenergy* 2, pp. 99–112.

Wicke B, Dornburg V, Junginger M, & Faaij A. 2008. Different palm oil production systems for energy purposes and their greenhouse gas implications. *Biomass & Bioenergy* 32(12):1322–1337.

Wilting H. An energy perspective on economic activities. PhD thesis. Groningen, 1996.

Zah R, H Böni, M Gauch, R Hischier, M Lehmann, & P Wäger. Ökobilanz von Energieprodukten: Ökologische Bewertung von Biotreibstoffen (Life Cycle Assessment of Energy Products: Environmental Impact Assessment of Biofuels). EMPA, St. Gallen, Switzerland, 2007.



Environmental Impact of Biofuels
Edited by Dr. Marco Aurelio Dos Santos Bernardes

ISBN 978-953-307-479-5
Hard cover, 270 pages
Publisher InTech
Published online 06, September, 2011
Published in print edition September, 2011

This book aspires to be a comprehensive summary of current biofuels issues and thereby contribute to the understanding of this important topic. Readers will find themes including biofuels development efforts, their implications for the food industry, current and future biofuels crops, the successful Brazilian ethanol program, insights of the first, second, third and fourth biofuel generations, advanced biofuel production techniques, related waste treatment, emissions and environmental impacts, water consumption, produced allergens and toxins. Additionally, the biofuel policy discussion is expected to be continuing in the foreseeable future and the reading of the biofuels features dealt with in this book, are recommended for anyone interested in understanding this diverse and developing theme.

InTech Europe
University Campus STeP Ri
Slavka Krautzeka 83/A
51000 Rijeka, Croatia
Phone: +385 (51) 770 447
Fax: +385 (51) 686 166
www.intechopen.com

InTech China
Unit 405, Office Block, Hotel Equatorial Shanghai
No.65, Yan An Road (West), Shanghai, 200040, China
中国上海市延安西路65号上海国际贵都大饭店办公楼405单元
Phone: +86-21-62489820
Fax: +86-21-62489821