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FACULDADE DE CIÊNCIAS E TECNOLOGIA

DEPARTAMENTO DE ENGENHARIA MECÂNICA

INCORPORATING UNCERTAINTY IN THE LIFE-CYCLE MODELING OF BIOFUELS: ENERGY RENEWABILITY AND GHG INTENSITY OF BIODIESEL AND BIOETHANOL IN EUROPE

João Manuel Nogueira Malça de Matos Ferreira

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M.M. 2011

Wheat is the most common cereal used for bioethanol production in Europe…

In the loving memory of my father

To my lovely daughters

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Contents

Abstract

Global warming and security of energy supply are main concerns in the international agenda. Renewable energy sources, including biofuels, are being promoted as possible contributions to address these problems. Nevertheless, significant disagreement and controversies exist regarding the actual benefits of biofuels displacing fossil fuels, as shown by a large number of publications that analyze the life-cycle of biofuels and that have varying and sometimes contradictory conclusions, even for the same biofuel type and pathway. A comprehensive assessment of the key issues that cause uncertainty and variability of the results is thus needed to ensure reliable outcomes and guarantee the environmental sustainability of policies and regulations at this level. Against this background, this dissertation aims to respond to the following questions: How do different (and alternative) European biofuel production systems compare each other? Are they equally efficient in terms of energy balance, GHG intensity, and land use? And how do biofuels compare with the fossil fuels they displace? These issues are assessed by providing a thorough review on European biofuel systems from an energy and greenhouse gas (GHG) life-cycle perspective, including detailed assessment of relevant aspects, namely data used, major assumptions, modeling choices and the extent to which they influence the results. The main sources of uncertainty impacting the life-cycle of biofuels are investigated and a robust framework for incorporating uncertainty issues in the modeling is implemented. To demonstrate the application of the methodology, life-cycle models for five European first-generation biofuel systems are developed, explicitly addressing uncertainty. Finally, the benefits and drawbacks of European biofuel systems –modeled under uncertainty– are presented and discussed in terms of energy renewability efficiency and GHG intensity. It has been concluded that: land use change dominates the GHG intensity of biofuels, although with a high level of uncertainty; optimum use of co-products is required to improve the energy efficiency and GHG intensity of biofuels; conversion of full- or low-tillage croplands to energy crops results in biofuel life-cycle GHG emissions lower than equivalent fossil fuel emissions,

whereas conversion of (improved) grassland to energy crops does not contribute to GHG savings over fossil fuels in the short- to mid-term. Calculated carbon payback times of the latter option are always above 50 years for the majority of biofuel systems.

As a closing remark for this dissertation:

Robust life-cycle modeling approaches incorporating uncertainty are essential to improve the transparency and reliability of life-cycle studies and better support decisions on whether or not to support specific biofuel pathways.

Sinopse

O aquecimento global e a segurança do abastecimento energético constituem importantes preocupações na agenda internacional. Diversas fontes de energia renovável, incluindo biocombustíveis, estão a ser promovidas como possíveis contribuições para solucionar estes problemas. No entanto, existe discordância e controvérsia acerca dos reais benefícios associados à substituição de combustíveis derivados do petróleo por sistemas de biocombustíveis, conforme vem sendo demonstrado num elevado número de publicações na literatura científica. Estes estudos, em que o ciclo de vida de diferentes cadeias de biocombustíveis é investigado, têm frequentemente conclusões diversas e, por vezes, contraditórias, inclusivamente para o mesmo tipo de biocombustível. Neste contexto, é fundamental uma avaliação integrada das principais causas subjacentes à incerteza e variabilidade nos resultados de diferentes estudos, de forma a garantir resultados e conclusões robustos e transparentes e, adicionalmente, garantir a sustentabilidade ambiental das políticas e regulamentações neste domínio. Esta dissertação pretende abordar estas questões sob uma perspectiva de ciclo de vida – limitada aos aspectos energético e de emissões de gases de efeito de estufa (GEE) – aplicada a sistemas de produção de biocombustíveis de primeira geração no contexto Europeu. Esta abordagem inclui a avaliação detalhada das principais fontes de incerteza afectando o ciclo de vida, nomeadamente dados utilizados, hipóteses metodológicas e opções de modelação assumidas. Para demonstrar a aplicação da metodologia, são desenvolvidos diversos modelos de ciclo de vida, abordando explicitamente a incerteza, para sistemas de biocombustíveis de primeira geração. São apresentadas e discutidas as vantagens e desvantagens de cinco sistemas europeus de biocombustíveis, modelados sob incerteza, em termos de eficiência de renovabilidade energética e de emissões de GEE. As principais conclusões do estudo indicam que: a mudança no uso do solo *(land use change)* constitui um factor determinante no balanço de gases de efeito de estufa dos sistemas de biocombustíveis, embora com um nível de incerteza elevado; é necessário um óptimo aproveitamento dos co-produtos, por forma

a melhorar a eficiência em termos energéticos e de emissões das cadeias avaliadas; a conversão de solos cultivados para os cultivos energéticos analisados nesta dissertação resulta em emissões de GEE no ciclo de vida dos biocombustíveis inferiores às dos correspondentes combustíveis derivados do petróleo; a conversão de prados com gestão melhorada para os cultivos energéticos analisados não contribui, no curto ou médio prazo, para poupanças de emissões de GEE no ciclo de vida relativamente aos combustíveis fósseis equivalentes.

Como conclusão final, esta dissertação demonstra que a modelação do ciclo de vida de biocombustíveis incorporando incerteza é essencial para promover a transparência e robustez dos estudos de ciclo de vida de biocombustíveis, contribuindo para melhorar a tomada de decisão neste domínio.

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1. Introduction

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1. INTRODUCTION

1.1. MOTIVATION

Global warming and security of energy supply are main concerns in the international agenda. Renewable energy sources, including biofuels, are being promoted as possible contributions to address these problems (EPC 2009; EPA 2010). Nevertheless, significant disagreement and controversies exist regarding the actual benefits of biofuels displacing fossil fuels, as shown by a large number of publications that analyze the life-cycle (LC) of biofuels and that have varying and sometimes contradictory conclusions, even for the same biofuel type and pathway (Farrell et al. 2006; Larson 2006; von Blottnitz and Curran 2007; Cherubini et al. 2009; Gnansounou et al. 2009; Luo et al. 2009; Malça and Freire 2010a; van der Voet et al. 2010; Börjesson and Tufvesson 2011). This stresses the need to identify the main drivers and to improve the knowledge of the sources for the differences and variations between different studies (and also within specific studies). Are they due to different methodological procedures (or modeling choices), data or production conditions?

Several issues have been found to affect the calculation of energy and greenhouse gas (GHG) balances of biofuels, namely: (i) modeling assumptions (e.g. approaches used for dealing with biofuel co-products; system boundaries and functional unit; consideration of a reference system for land use); (ii) model simplifications in comparison to real world systems (model uncertainty); (iii) data quality for key input parameters (e.g. fertilizers and fuel used during raw material cultivation; soil emissions due to land use and land use change); and (iv) type of indicators used to communicate the results (Cherubini et al. 2009; Gnansounou et al. 2009; Cherubini 2010; de Vries et al. 2010; Hoefnagels et al. 2010; van der Voet et al. 2010; Whitaker et al. 2010; Malça and Freire 2011a).

Even though a few life-cycle assessment (LCA) studies take into account uncertainty and variability issues, namely through scenario analysis, single parametric sensitivity analysis or Monte-Carlo simulation, they usually consider the treatment of uncertainty as an appendix, and only consider portions of the uncertainty instead of focusing on the overall uncertainty itself (Plevin 2010; Malça and Freire 2011a). Moreover, several review studies show that important aspects for the GHG balance of biofuels, namely direct and indirect soil carbon emissions from land use change (LUC), have not captured enough attention, even in recent biofuel life-cycle studies (de Vries et al. 2010; van der Voet 2010; Whitaker et al. 2010; Malça and Freire 2011a). Nevertheless, indirect LUC associated to biofuels has recently been the subject of important controversy among the scientific community and further work is still required to address the practical modeling of the issue (Anex and Lifset 2009; Kløverpris et al. 2008a). Therefore, indirect LUC is beyond the scope of this dissertation.

The LCA methodology and a comprehensive assessment of key issues that cause uncertainty and variability in biofuel life-cycle studies are central in searching for the best biofuel routes concerning energy and GHG balances. Through this framework it is possible to identify critical issues within biofuel chains, enabling opportunities for optimization and overall improvement of the system. Moreover, coupling LCA and uncertainty assessment provides a sound basis to compare alternative biofuel systems and assess the potential benefits of biofuels over fossil fuels.

In short, a comprehensive assessment of the key issues that cause uncertainty and variability in biofuel life-cycle studies is needed to ensure reliable outcomes, promote better practices and preference for improved biofuel options, in order to guarantee the environmental sustainability of European policies and regulations at this level.

1.2. OBJECTIVES

The main objective of this dissertation is to respond to the following questions:

How do different (and alternative) European biofuel production systems compare each other? Are they equally efficient in terms of energy balance, GHG intensity, and land use? And how do biofuels compare with the fossil fuels they displace?

As stated in the previous section, robust life-cycle modeling approaches incorporating uncertainty are essential to improve the transparency and reliability of biofuel life-cycle studies and better support decisions on whether or not to support specific biofuel pathways. In this context, the dissertation addresses the following research questions:

I. What drives the differences and sometimes contradictory conclusions between life-cycle studies, even for the same biofuel pathway?
II. How to develop life-cycle models for biofuel systems incorporating uncertainty?
III. How uncertain are the energy and GHG emission results from European biofuel (biodiesel and bioethanol) life-cycle studies?
IV. Given the uncertainty ranges, is it possible to ensure that biofuels are really delivering energy and GHG savings over displaced petroleum fuels? And to what extent?
V. What direction should research take to improve the robustness of biofuel life-cycle studies?

1.3. CONTRIBUTION

The main contributions of this dissertation can be synthesized as follows:

1. A novel metric is proposed for quantification of the energy renewability of (bio)fuel systems, the *Energy Renewability Efficiency* (ERenEf). Within life-cycle energy studies focusing on (bio)fuel systems, the adequacy of this metric is manifest, as it emphasizes the merits of (bio)fuels from the standpoint of renewable energy usage as opposed to the more conventional approach of non-renewable energy input estimation of other metrics;
2. A thorough review is conducted on biofuels in Europe from an energy and GHG life-cycle perspective, including detailed assessment of relevant aspects, namely data used, major assumptions, modeling choices and the extent to which they influence the results;
3. The main sources of uncertainty impacting the life-cycle of biofuels are investigated and a robust framework for incorporating uncertainty issues in the modeling of biofuel systems is proposed. To demonstrate the application of the methodology, life-cycle models for several European first-generation biofuel systems explicitly addressing uncertainty are developed;
4. Finally, the benefits and drawbacks of European biofuel systems –modeled under uncertainty– are presented and discussed in terms of energy renewability efficiency and GHG intensity. Energy and GHG savings over petroleum fuels are also calculated.

Some of these contributions have already been published in ISI journals, as listed below:

- 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.

In addition, more than twenty publications in conference proceedings and book chapters have been authored in the field, of which a selection is included in the "Bibliographic References" section. At present, two additional manuscripts are being prepared for journal submission, in which the influence of land use change and cultivation practices in European biofuel (biodiesel and bioethanol) production systems is thoroughly analyzed.

1.4. STRUCTURE OF THE DISSERTATION

The formulated research questions are addressed throughout Chapters 3 to 7 of this dissertation, as shown in Table 1.1. The structure of the dissertation is described in the following paragraphs.

Table 1.1. Overview of research questions and chapters in which they are addressed.

	Research Questions				
Chapter	**I**	**II**	**III**	**IV**	**V**
3	✔		✔	✔	
4		✔			
5			✔		
6			✔	✔	
7	✔	✔	✔	✔	✔

Following this introduction, chapter 2 provides a background on uncertainty assessment, describing the main typologies of uncertainty and the approaches used to manage and incorporate different types of uncertainty in the life-cycle of biofuels. Moreover, chapter 2 discusses the importance of several methodological issues in the Life-Cycle Assessment (LCA) of biofuel pathways and reviews the metrics commonly used in the life-cycle

energy and GHG assessment of biofuel systems. Chapter 2 is thus a background chapter for the remainder of the dissertation.

Chapter 3 presents a comprehensive review of biofuel life-cycle studies in Europe, identifying the major aspects that motivate divergences and sometimes contradictory conclusions between studies. A detailed assessment of the major assumptions and modeling choices in each surveyed study and how they affect life-cycle results (non-renewable primary energy requirement and GHG intensity) is presented.

Chapter 4 describes representative European biofuel systems, including life-cycle models for five first-generation biofuel pathways: Rapeseed Oil and its derivative Rapeseed Methyl Ester, Bioethanol from Wheat, Bioethanol from Sugar Beet, and the bioethanol derivative Bio-Ethyl Tertiary Butyl Ether.

Chapter 5 discusses modeling issues concerning soil carbon fluxes from direct land use change. This chapter presents several approaches for evaluating soil carbon stock exchange associated with LUC and their implications in the life-cycle GHG balance of biofuel systems are discussed.

Chapter 6 characterizes biofuel chains in terms of energy renewability efficiency and GHG intensity, enabling comparison between biofuel systems and also against displaced petroleum fuels. Several biofuel modeling aspects with potential implications in the results are also explored in this chapter.

Chapter 7 concludes the dissertation with the main findings of the research and an examination of the main strengths and limitations of the investigation. This chapter draws the conclusions concerning the research questions of the dissertation and points out important recommendations for future research.

2. Addressing Uncertainty in the Life-Cycle Modeling of Biofuels

"Very possibly, we may even be uncertain about our degree of uncertainty. The variety of types and sources of uncertainty, along with the lack of agreed terminology, can generate considerable confusion."

Morgan and Henrion (1990)

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2. ADDRESSING UNCERTAINTY IN THE LIFE-CYCLE MODELING OF BIOFUELS

2.1. PURPOSE AND SCOPE

Underlying the core of this dissertation, which is to capture uncertainty issues in the life-cycle energy renewability and GHG intensity of biofuel systems, two important concepts emerge: (i) the assessment of *uncertainty*; and (ii) the use of a *life-cycle perspective.* On the one hand, the variety of types and sources of uncertainty and the lack of agreed terminology may give rise to diverging and contradictory conclusions between biofuel studies. On the other hand, the methodology of Life-Cycle Assessment (LCA) holds in itself important methodological challenges, which add to the difficulties of dealing with uncertainty. This broad range of concepts is addressed in this chapter, setting the grounds for the remaining chapters of the dissertation.

2.2. DEALING WITH UNCERTAINTY

Common practice in life-cycle studies includes the modeling of product systems in order to quantify the resource consumption, energy and environmental impacts over the life-

cycle. Most often, LCA practitioners build deterministic models to approximate real systems and thus fail to capture the uncertainty and variability inherent in LCA (Lloyd and Ries 2007). This type of approach results in incomplete outcomes that may be erroneously interpreted, or worse, may promote decisions in the wrong direction (Lloyd and Ries 2007; Plevin 2010). In order to promote LCA as a reliable decision tool it is thus important that uncertainty and variability are taken into account in life-cycle modeling (Williams et al. 2009). Several techniques and tools for conducting life-cycle studies under uncertainty have been proposed and implemented (see e.g. Björklund 2002; Heijungs and Huijbregts 2004; Lloyd and Ries 2007).

Uncertainty analysis assumes particular relevance in the sustainability assessment of biofuels, where comparisons with fossil fuel equivalents in searching for effective GHG emission reductions are on top of the international agenda (EPC 2009; EPA 2010). However, though many studies have addressed the environmental life-cycle impacts of biofuels, they often neglect data uncertainty assessment and the implications of modeling assumptions (Malça and Freire 2011a).

2.2.1. Typology of uncertainty

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 (ISO 14040, 14044:2006). 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; Huijbregts 2001; 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. A first broad distinction is usually made between uncertainty and variability. The former relates to a lack of knowledge: no data is available, or the available data is wrong or ambiguous. Variability, in contrast, is a

quality of data that is heterogeneous, and changes across time, space or individuals (Heijungs and Huijbregts 2004; Krupnick et al. 2006). Surveys on possible ways of typifying uncertainty and variability can be found, for example, in Huijbregts (1998), Björklund (2002), Heijungs and Huijbregts (2004), Lloyd and Ries (2007), and Williams et al. (2009).

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.

As noted by Morgan and Henrion (1990), the distinction between uncertainty about model structure (model uncertainty) and uncertainty about quantity values (parameter uncertainty) is rather slippery: when a model parameter is defined to select among different functional forms, the modeler is actually converting model uncertainty into parameter uncertainty (Morgan and Henrion 1990; Plevin 2010). On the other hand, uncertainty due to choices is unavoidable in LCA and depends on the options taken by the modeler (Björklund 2002). There is often not one single correct choice; thus, a sensitivity analysis for different alternatives should be conducted (Björklund 2002; van der Voet et al. 2010).

Concerning variability, a classification into three categories is proposed in the literature (see e.g. EPA 1997; Huijbregts 1998; Björklund 2002; Geisler et al. 2005):

- spatial (or geographical) variability, which accounts for variations across locations. This type of variability can occur at different levels, namely regional

(e.g. typical fertilizer application rates in different regions within a country) and local (e.g. differences between production sites) levels;

- temporal variability, which refers to variations over time, whether long- or short-term (e.g. variation of agricultural yields across years); and
- variability between sources and objects, which may occur due to differences between sources (e.g. inherent variations in comparable technical processes) or between objects (e.g. variability in human characteristics and related sensitivity to specific environmental impacts).

Despite the differences at a more fundamental level between uncertainty and variability, the approaches for dealing with the two show a large overlap (Heijungs and Huijbregts 2004) and can be addressed using the same techniques (Notten and Petrie 2003; Geisler et al. 2005). For this reason, the word “uncertainty” is sometimes used interchangeably for both uncertainty and variability, see e.g. Björklund (2002) and Lloyd and Ries (2007). As emphasized in Krupnick et al. (2006) and Plevin (2010), the importance of typifying uncertainty is somehow secondary. Actually, the main purpose of developing a taxonomy for uncertainty is to guide LCA practitioners and decision-makers to identify and consider all sources of uncertainty.

Lloyd and Ries (2007) conducted a comprehensive survey of 24 publications addressing quantitative uncertainty analysis in LCA. This review shows that, 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. Important recommendations for life-cycle studies were drawn from this survey, namely:

- the need to conduct uncertainty importance analysis in order to highlight areas in which an improved understanding is required;
- the need for a better understanding of the importance of different types of uncertainty and variability.

Details concerning the assessment of uncertainty are provided in the following sections.

2.2.2. Parameter Uncertainty

Every type of modeling is associated with uncertainties in its parameters (Schade and Wiesenthal 2011). Methods for propagating parameter uncertainty in life-cycle studies include stochastic modeling (e.g. Geisler et al. 2005; Peters 2007; Soimakallio et al. 2009), fuzzy data sets (Tan et al. 2002; Tan 2008), interval calculations (Chevalier and Téno 1996), Bayesian statistics (Lo et al. 2005; Johnson 2006), analytical uncertainty propagation (Heijungs 1996; Hong et al. 2010), and combining approximation formulas and Monte Carlo simulation (Ciroth et al. 2004). A survey of the merits and limitations of these techniques is presented in Lloyd and Ries (2007).

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.

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.

CHOICE OF PROBABILITY DISTRIBUTIONS. A challenging aspect in stochastic modeling is the assignment of probability distributions to parameters, a task that depends on the amount of available information. Techniques for choosing a probability distribution can

be summarized as follows (Björklund 2002; Landis et al. 2007; Lloyd and Ries 2007; Plevin 2010): (i) for parameters with large data sets, statistical analysis can be used for curve fitting and computation of the best-fit probability distribution[1]; (ii) small sample sizes can be represented by frequency distributions (histograms), in which data is organized in classes and frequencies, and that can be later approximated by probability distributions; (iii) when limited information is available, a uniform distribution can be assigned (for parameters with single or two data points), or, if one value appears to be more likely, a triangular distribution can be used. When uncertainty is subjective and statistical analysis is not possible, a best estimate can be used based on the experience of an expert in the field – expert judgment (Björklund 2002; Lloyd and Ries 2007).

CORRELATED VARIABLES. If correlation between parameters is identified, it can be incorporated in a Monte-Carlo simulation to restrain the choice of possible values assigned to parameters. This approach avoids taking into account meaningless scenarios in the model (Schade and Wiesenthal 2011). If correlations are difficult to assess, it is possible to structure the model so that the correlations are represented internally (Plevin 2010). These are the two primary approaches to incorporating dependencies in the model structure (Cullen and Frey 1999; Krupnick et al. 2006). Nevertheless, correlation among inputs may not always be important to the assessment of uncertainties, except if the output uncertainty is sensitive to the correlated inputs and the correlation is sufficiently strong (IPCC 2006).

2.2.3. Uncertainty due to Choices (Scenario Uncertainty)

MULTIFUNCTIONALITY. 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

[1] Several tests are available for best-fit regression, e.g. the Chi-squared test.

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 ambiguity on how to allocate input requirements between outputs in joint production[2] had already been raised in the early development stages of energy analysis[3]. According to the recommendations set forth in the workshop report, energy requirements should be partitioned according to a physical parameter whenever possible. In the case of fuel products, "it would be natural to partition energy inputs according to the energy embodied in the various outputs", whereas partition according to product money values was suggested for some policy applications (Long 1978). Moreover, reporting of total unpartitioned requirements was also recommended, so that other analysts accessing the data were able to use their own partitioning schemes.

According to ISO 14044:2006, the options for dealing with co-production include: (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 partitioning (allocation) methods are straightforward to implement, they "arbitrarily" allocate inputs and outputs on the basis of specific relationships between co-products (Weidema et al. 2003). For this reason, ISO standards on LCA indicate that allocation 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).

[2] In *joint production*, the share of co-product outputs is fixed. In contrast, the output of co-products can be varied independently in *combined production* (Frischknecht 2000).

[3] At the 1975 IFIAS workshop (Long 1978). The International Federation of Institutes for Advanced Studies (IFIAS) was established in 1972 as a potential new instrument for truly transdisciplinary and transnational cooperation in the physical, biological and social sciences and the humanities (Roberts 1975).

Weidema and Schmidt (2010), for example, use the expression “system expansion”, but they are actually referring to the substitution method. 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. Another example of language imprecision comes from the meaning of allocation in LCA, which is often used misleadingly. With this respect, see e.g. Bernesson et al. (2004) and van der Voet et al. (2010) which classify the substitution approach as an additional allocation method. According to ISO 14044:2006, sub-division and system boundary expansion are not formally part of the allocation procedure.

A complete subdivision is not possible in joint production, due to the fixed share between co-product flows. It is only feasible in sub-processes that are separate in space and/or time, i.e. in combined production (Gnansounou et al. 2009). Ekvall and Finnveden (2001) analyzed a large number of LCA studies where subdivision or system expansion was applied and found no case study where an allocation problem is completely eliminated through sub-division. Moreover, it has generally been regarded as impossible to expand a system in all cases (Weidema et al. 2003). System expansion requires that there is an alternative way of generating the exported functions and that data can be obtained for this alternative production (Tillman 2000; Ekvall and Finnveden 2001). Many co-products are competing with other co-products, so expanding the system boundary would only result in an increasingly complex system (Beer et al. 2001; Elsayed et al. 2003). In particular, many of the co-products of biofuel technologies have no separate main means of production. Hence, a simple substitute cannot be identified. Another difficulty arises when the market for the most realistic replacement is restricted, which requires the coupling of system expansion to a specific amount of biofuels produced (Börjesson and Tufvesson 2011). Two illustrative examples of the additional complexity in using system expansion and the substitution method can be found, respectively, in Kim and Dale (2002) and Reinhard and Zah (2009). A straightforward alternative to this complexity is to use an allocation method.

According to the ISO 14044:2006, allocation should reflect the physical relationships between the environmental burdens and the functions, i.e. how the burdens are changed by quantitative changes in the functions delivered by the product system. Thus, 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; Harding 2007; Neupane 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. Moreover, 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 is indicated for the regulation of individual economic operators, because it is easy to apply, is predictable over time and minimizes counter-productive incentives (EPC 2009). Nevertheless, this approach is not the most appropriate when biofuel co-products are not themselves energy products, as is often the case.

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 (2009). 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 (Bergsma et al. 2006; 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 2006a; 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 is rather easily illustrated by identifying the relevant alternatives and performing sensitivity analysis, as mentioned before.

UNCERTAINTY OF GLOBAL WARMING POTENTIALS. Several time horizons can be adopted for the estimation of GHG emissions, namely 20, 100, and 500 years (IPCC 2007). Most commonly, a time horizon of 100-years is used (Levasseur et al. 2010; Plevin 2010). Sections 2.3.1 and 2.3.3 present more details.

2.2.4. Difficulties and Limitations

As stated by Plevin (2010), assessment of uncertainty issues is often completely avoided or at least performed in a limited manner. Incorporating uncertainty can add significant complexity to a model and performing an uncertainty analysis can be very time-consuming. Moreover, the results of a modeling study incorporating uncertainty may be less clear than hoped. This raised concerns on some LCA practitioners and experts that high levels of uncertainty and the resulting lack of significance between outcomes turned results of life-cycle studies meaningless (Huijbregts et al. 2003). Ekvall and Weidema (2004) also raise the same concerns.

On the other hand, ignoring uncertainty can give a false impression of distinguishability among alternatives (Basson and Petrie 2007). An example brought from the sustainability assessment of biofuels is the conclusion in several studies that the allocation method for treating co-products largely influences the life-cycle results. Does this conclusion hold when parameter uncertainty is included in the assessment? Therefore, even if abilities are limited, a preferred approach is to conduct uncertainty analysis as best as possible: “almost any uncertainty analysis is better than none at all” (Krupnick et al. 2006, p.8).

An additional difficulty when managing all types of uncertainty simultaneously is the damping effect that the introduction of allocation coefficients has on the computed parameter uncertainty ranges of the outputs. This results in artificially narrow uncertainty ranges for output values. On the other hand, when the substitution method is used, not only the overall uncertainty of the biofuel system is taken into account in the output values but also the uncertainty associated to the products displaced by biofuel co-products.

One of the issues raised by Lloyd and Ries (2007) in their survey of approaches to incorporate quantitative uncertainty into LCA was the need to maintain conservation of mass and energy. ISO standards on LCA (ISO 14044:2006) also point out calculation of mass and energy balances as an additional check on data validity. However, as demonstrated by Weidema and Schmidt (2010), the substitution method always ensures mass and energy balances, whereas allocation may certainly not. In the substitution

method, the resulting system originates from subtracting avoided processes to the process of interest. Therefore, if the affected unit processes have correct physical balances, the same occurs with the resulting system in the substitution method. In contrast, allocation breaks up the system under study into two or more artificial systems, based on an arbitrary allocation rule (e.g. mass, energy, market price). In this case, the only physical balance that remains correct is that given by the allocation key, whereas other balances most certainly become skewed (Weidema and Schmidt 2010).

An additional advantage of the substitution method (and also system expansion) over allocation methods is that in the former all significant processes affected by the system under study are included in the analysis, namely by a change in the amount of co-products (ISO 14044:2006; Weidema and Schmidt 2010).

2.3. LIFE-CYCLE ASSESSMENT OF BIOFUELS

In addition to the difficulties and limitations of addressing uncertainty, conducting a life-cycle assessment is also subject to several methodological challenges, as stated in the beginning of the chapter. A Life-Cycle Assessment study offers a comprehensive picture of the flows of energy and materials through a system and gives a holistic and objective basis for comparison. LCA 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 should provide a full and definitive description of the product or service being investigated, enabling subsequent results to be interpreted 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, performing allocation steps for multifunctional processes and completing the final calculations (Guinée et al. 2002). 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 focused in specific environmental impact categories. Most common types of studies include (Liska and Cassman 2008; Cherubini and Strømman 2011): (i) life-cycle energy analysis, which is limited to assess energy aspects and, in this dissertation, with a particular focus on energy efficiency indicators aiming at characterizing the renewability of biofuel systems (see section 2.3.2 for further details); (ii) life-cycle GHG assessment, in which the GHG balance of the biofuel system over the entire life-cycle is quantified (see section 2.3.3); and (iii) life-cycle assessment, in which a set of environmental impact categories are investigated.

Van der Voet et al. (2010) present a subdivision concerning the particular purpose of biofuel LCA studies. These authors consider three different fields of application:

- 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.

Regardless of the goal and scope of a biofuel life-cycle study, important methodological challenges within the field of biofuel LCAs can be identified.

2.3.1. Methodological Issues in Life-Cycle Studies of Biofuels

According to Reap et al. (2008a), Life-Cycle Assessment is a methodology in need of improvement. These authors conducted a comprehensive literature survey in which several problems and difficulties throughout all LCA phases are identified (Reap et al. 2008a, 2008b). In the following paragraphs, methodological issues concerning the application of life-cycle approaches in the sustainability assessment of biofuels are explored (Fig. 2.1). These issues may have important implications in the results of a life-cycle study, as demonstrated in the remaining chapters of this dissertation.

Fig. 2.1. Methodological challenges affecting the results of biofuel life-cycle studies.

FUNCTIONAL UNIT. The definition of a functional unit is an important step in a Life-Cycle Assessment (Matheys et al. 2007; 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 (Weidema et al. 2004; ISO 14040:2006; Matheys et al. 2007).

The definition of the functional unit is related with 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 2011a):

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

To ensure comparability between different biofuel life-cycle studies, results should be expressed in terms of the same functional unit, which in turn ensures that the comparison is based on delivering the same service (Heijungs et al. 1992; Weidema et al. 2004; ISO 14040:2006). Nevertheless, as long as system boundaries are appropriately set and additional data is provided, results of life-cycle studies with different functional units can be easily recalculated to match each other (van der Voet et al. 2010).

The adoption of delivered fuel energy (e.g. 1 MJ) as the functional unit avoids the complexities of adding further assumptions, in particular concerning vehicle performance factors and selection of driving cycles, as it would be if, for example, distance traveled was adopted as the reference. Some authors argue that this approach should be the first-choice in life-cycle studies of biofuels, since different fuels may have different engine energy efficiencies and, thus, should be compared for the same transportation service (Cherubini et al. 2009; Gnansounou et al. 2009). The additional difficulty of this approach can be avoided assuming that the biofuels under investigation have similar combustion characteristics compared to conventional fossil fuels (Hoefnagels et al. 2010). This justifies the much straightforward use of an energy-oriented functional unit or, alternatively, the use of a predetermined conversion factor from energy to a per vehicle-km basis. The correct approach, however, is to include in the calculations the improved/reduced efficiency of biofuels over fossil fuel comparators on the basis of field tests (see e.g. Gnansounou et al. 2009 and Lechón et al. 2009). Adding to the complexity of this approach is the potential high range of uncertainty that may be introduced, as shown in Hekkert et al. (2005).

Other biofuel life-cycle studies use agricultural surface area – usually on a per hectare basis – as the functional unit, which is motivated by the limitation in terms of available area for growing energy crops (Cherubini et al. 2009). The option for mass- or volume-

based functional units is also used in several studies (e.g. Shapouri et al. 1995; Kim and Dale 2002; Shapouri et al. 2002; Henke et al. 2005). However, this is definitely not an adequate basis for comparison of the function provided by different (bio)fuels. To note that the selection of a particular functional unit influences the energy and GHG metrics to be used in the life-cycle study (see sections 2.3.2 and 2.3.3).

SYSTEM BOUNDARIES. The definition of system boundaries in a life-cycle study is related to the functional unit of the study and determines which processes are included in the assessment[4]. If boundaries include the very upstream process up to the final product, the approach is often designated as cradle-to-gate. The "gate" can be seen here as the fuel pumping station where biofuel is delivered to vehicles. The cradle-to-gate approach is often called well-to-tank (WtT) approach in transportation systems literature and, to a lesser extent, well-to-pump (Huo et al. 2009) or seed-to-tank (Reijnders 2009) assessment. Several biofuel life-cycle studies take into account the biofuel use phase. This component of the study is the tank-to-wheels (TtW, also called pump-to-wheels) assessment, and covers only the vehicle operation activities, which can be based on data from vehicle simulation models, on-road testing, engine dynamometer experiments or fleet operation data. A well-to-wheels (WtW) modeling boundary includes both the well-to-tank and tank-to-wheels stages.

It must be emphasized that the shift from the WtT to the WtW perspective may change life-cycle results, because compression-ignition (CI) engines have in general higher thermal efficiencies compared to spark-ignition (SI) engines. This is particularly sensitive when the same fuel is used in different types of engines (e.g. bioethanol can be used either in SI or CI engines) or when different fuels are compared based on several functional units. In these cases, comparison data and results must be analyzed carefully.

The selection of the system boundary shall be consistent with the goal of the study. The WtT assessment is particularly appropriate if the goal and scope is concerned with biofuel use as a generic energy carrier, without a particular transportation or energy conversion system being considered. The WtT assessment enables life-cycle inventory

[4] Cut-off criteria are typically employed to reduce data collection efforts in the LCI phase (ISO 14040:2006; Plevin 2010), although this approach introduces a truncation error in the assessment (Suh et al. 2004).

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 addressing uncertainty.

Other important issues concerning the definition of system boundaries are the inclusion of the reference system to which the biofuel system is to be compared (see next paragraph), as well as consideration of potential substitutes for co-products when the biofuel chain is multifunctional. This last aspect has already been discussed in section 2.2.3.

REFERENCE SYSTEM. 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) in the European Union and EPA (2010) in the USA). Most commonly, the reference system used is a fossil fuel pathway (gasoline or diesel). Less common alternatives include non-conventional sources of liquid fossil fuels, such as oil from tar sands or Fischer-Tropsch diesel from coal (Hoefnagels et al. 2010). The EU directive 2009/28/EC (EPC 2009), for example, has adopted a generic reference value for life-cycle GHG emissions of fossil fuels used for transportation (83.8 g CO_2eq MJ^{-1}), not distinguishing between gasoline 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.

TIME FRAME. Different choices concerning time-dependent aspects in biofuel life-cycle studies may impact the final results. This dissertation addresses first-generation biofuel systems in Europe and thus it is devoted to the annual life-cycle performance of biofuel systems currently being produced at commercial scale. Two important time-related issues include

- the time frame chosen for annualization of soil emissions from land use change (see section 2.3.3), which may have a strong impact in the results (Hoefnagels et

al. 2010). Some authors use a default value of 20 years (IPCC 2006; BSI 2008; EPC 2009), whereas other authors admit longer periods (e.g. 30-yrs in Searchinger et al. 2008, 100-yrs in Kim et al. 2009) or shorter periods (10-yrs is demanded by Greenpeace, according to Croezen and Kampman 2008);

- the time horizon selected for estimation of the global warming effect of greenhouse gases. The time horizon of 100 years is generally accepted, because it reflects the approximate time carbon dioxide molecules (CO_2 is the reference gas for global warming calculations) remain in the atmosphere (Plevin 2010). The Kyoto Protocol has also settled on 100-year GWPs for national GHG inventories (Plevin 2010). Although the choice of a particular time horizon is ultimately arbitrary in a life-cycle assessment (Jackson 2009), GHG reductions in the short-term are crucial to avoid irreversible adverse effects from climate change (Searchinger et al. 2008).

TYPE OF LCA APPROACH. Two different approaches to LCA have been proposed (ISO 14040:2006): attributional (or retrospective) life-cycle assessment (ALCA) and consequential (or prospective) life-cycle assessment (CLCA). The attributional approach for LCA aims at describing environmentally relevant physical flows to and from a life-cycle and its sub-systems and therefore uses average data. Prospective or consequential LCA aims at assessing the consequences of change compared to the present situation, that is how the environmentally relevant physical flows to and from the life-cycle will change in response to possible changes; therefore, consequential LCA uses marginal data (Ekvall and Andrae 2006; Schmidt 2008). A few studies of CLCA applied to biofuel systems have been recently published (Schmidt 2008, 2010; Reinhardt and Zah 2009, 2011; Kløverpris et al. 2010; Overmars et al. 2011).

Nonetheless, CLCA is still object of much research and debate. Consensus on when to use CLCA and standardizing the CLCA procedure are still under development (Earles and Halog 2011) and a reliable methodology has yet not been established for bioenergy studies (Anex and Lifset 2009). An aspect that requires a consequential approach in life-cycle studies is the assessment of indirect land use change associated with biofuels, which will be discussed in the next paragraph.

INDIRECT LAND USE CHANGE (iLUC). Increased biofuels 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 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 for several first and second generation biofuels. This report shows that overall emissions from iLUC are within 10 to 80 g CO_2 MJ^{-1} of biofuel produced. Kløverpris et al. (2008b, 2010) also used a dynamic economic model to estimate long-term land use consequences of changes in crop consumption and exemplified their approach for wheat cultivation in different regions of the world.

Instead of economic models, other authors simplistically assume that a single marginal supplier and marginal product can be identified (Kløverpris et al. 2008b; Plevin 2010), which is valid if only small changes in the world´s market basket are induced by the biofuel system under analysis (Plevin 2010). For example, Reinhard and Zah (2009) conducted a CLCA study for Switzerland to estimate the environmental impacts of replacing 1% of fossil diesel consumption by biodiesel imported from Brazil or Malaysia.

Other attempts for addressing indirect land use change and its influence on life-cycle results use single CO_2 emission factors –the iLUC factor approach (e.g. Bowyer 2010; Fritsche et al. 2010; Overmars et al. 2011). 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) and Liska and Perrin (2009), so that a harmonized methodology can be established. In a report published on December

2010 (EC 2010d), the EU recognizes that a number of uncertainties associated with iLUC modeling remain to be addressed, which could significantly impact the results. A detailed assessment of potential policy approaches for dealing with iLUC issues is currently being conducted. Main conclusions from this assessment and legislative proposals, if appropriate, for amending the renewable energy and fuel quality directives are expected in the near future (EC 2010d). Due to the abovementioned difficulties and the lack of an agreed methodology, indirect LUC is not explicitly incorporated in this dissertation.

CAPITAL GOODS. Many LCA studies exclude per se the contribution of capital goods in the life-cycle inventory of a product or service (Frischknecht et al. 2007). However, according to the ISO standards, energy and emissions associated with facilities and machinery used in a product system are explicitly part of the system (ISO 14040, 14044:2006). As such, cut-off criteria apply on capital goods as on any other input (Frikschnecht et al. 2007). In particular, if the contribution of capital goods is considered relevant after a previous estimation of its magnitude, the definition of the system boundaries of the study must reflect it, and vice-versa.

2.3.2. Life-Cycle Energy Analysis

Energy analysis was established in 1974 as a new field devoted to the evaluation of resource flows in societal processes. Following two IFIAS Workshops, in 1974 and 1975, convened to establish the basic ground rules for energy analysis, a definition was agreed: energy analysis is “the determination of the energy sequestered in the process of making a good or service within the framework of an agreed set of conventions, or applying the information so obtained” (Roberts 1975; Long 1978). Even at the early stages of the “energy analysis” field, the importance of the “energy costs of getting and concentrating” net energy was emphasized. The expression “energy sequestered” was used to “indicate that energy may be tied up in the finished good or in process materials, in addition to the energy used to do the work of the process” (Roberts 1975).

Energy resource depletion must be quantified in terms of primary energy, i.e. energy embodied in natural resources (e.g. coal, crude oil, uranium or biomass) that has not undergone any anthropogenic conversion or transformation. Primary energy values are an indicator of energy resource availability and implicitly take into account the energy quality. 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):

$$GER = E_{in,non\text{-}renewable,prim} + FEC \quad (2.1)$$

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.

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 final fuel. The more non-renewable energy is required to make the biofuel, the less we can say that this 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).

Another source for differences between studies within the energy analysis and LCA literature is the 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 (Basset et al. 2010; Börjesson and Tufvesson 2011); cumulative energy demand (Wagner and Pick 2004; 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; de Vries et al. 2010; Papong and Malakul 2010); net energy yield (Liska and Cassman 2008); and energy return on investment[5] (Hammerschlag 2006; 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:

$$LCEE = \frac{\mathrm{FEC}}{(\mathrm{E_{in,non-renewable,prim}} + \mathrm{FEC})} \tag{2.2}$$

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:

$$FER = \frac{\mathrm{FEC}}{E_{in,non-renewable,prim}} \tag{2.3}$$

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

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" (Elsayed et al. 2001; Elsayed et al. 2003; Mortimer et al. 2003; Malça and Freire 2004a, 2006a; Hoefnagels et al. 2010):

$$E_{req} = \frac{E_{in,non-renewable,prim}}{FEC} \quad (2.4)$$

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

$$NEV = FEC - E_{in,non-renewable,prim} \quad (2.5)$$

is used e.g. in Shapouri et al. (1995), Shapouri et al. (2002), Liska and Cassman (2008) and Papong and Malakul (2010)[6]. 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. In this dissertation, the energy requirement E_{req} is used to identify the relative contributions to the total primary energy input from different stages of the production chains and to evaluate the implications of the allocation method chosen for the energy efficiency of biofuels.

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

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

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 (Schlamadinger et al. 2005; 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.

In addition to the metrics mentioned above, a novel indicator has been proposed by Malça and Freire (2004a, 2006a): the *Energy Renewability Efficiency*, aiming at characterizing the renewability of an energy system. The energy renewability efficiency (ERenEf) – to our knowledge, not previously proposed in the literature – 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 2004a, 2006a). 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:

$$ERenEF\ [\%] = \frac{(\mathrm{FEC} - \mathrm{E}_{\mathrm{non-renewable,prim}})}{\mathrm{FEC}} \times 100 \qquad (2.6)$$

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, gasoline (the fossil fuel displaced by bioethanol) shows an average ERenEf value of –19.0%, meaning that the total primary energy required to produce gasoline is 19.0% greater than its final energy content.

2.3.3. Life-Cycle GHG Assessment

The methodology for calculating the GHG balance of biofuel systems is presented in this section. 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 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 is 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). Additional emissions are due to the manufacture of feedstock inputs, extraction, transportation and transformation of raw fossil fuels, and electricity generation. GHG emissions for feedstock and energy inputs are calculated by using suitable emission factors (Mortimer and Elsayed 2006; Malça and Freire 2010a).

For comparative and decision purposes, GHG emission savings can be calculated by comparing the life-cycle GHG emissions of biofuels with the GHG emissions of fossil fuels, following the methodology used e.g. in EPC (2009) and Bergsma et al. (2006):

$$GHG_{emission\ savings}\ [\%] = \frac{(\text{Fossil Fuel}_{\text{emissions}} - \text{Biofuel}_{\text{emissions}})}{\text{Fossil Fuel}_{\text{emissions}}} \times 100 \qquad (2.7)$$

Biofuel use (combustion in engines or boilers) is not explicitly modeled, but it is assumed that tailpipe CO_2 emissions from biofuel combustion are neutral, being balanced by the CO_2 sequestered during crop growth (biogenic CO_2), which does not occur for fossil fuels. Nevertheless, it must be emphasized that the non-renewable fraction in a biofuel[7], if any, contributes to net CO_2 emissions during the combustion phase, and must be properly accounted for.

[7] Being of non-renewable origin, this fraction in a biofuel contributes to emissions that were not previously offset and thus must be taken into account in the GHG assessment.

An alternative approach to excluding biogenic carbon is 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). In this case, biogenic carbon is first included as an extraction (due to photosynthetic sequestration of atmospheric CO_2) and later in the chain as an emission (during biofuel combustion).

DIRECT LAND USE CHANGE[8] AND LAND USE. The significant variability of published data concerning soil GHG emissions and the fact that many life-cycle studies do not take into account this issue may affect the credibility of LCA calculations (St. Clair et al. 2008). Soil carbon stock change, in particular, is an emergent topic in the literature and can contribute significantly to biofuel GHG intensity (Brandão et al. 2011). 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 2011a). 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} (tonnes per hectare per year, t C ha^{-1} yr^{-1}) are given by (EPC 2009)

$$\Delta C_{LUC} = \frac{CS_R - CS_A}{T_{LUC}} \quad (2.8)$$

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 actual 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

[8] Another aspect within the life-cycle GHG assessment of biofuel systems is indirect land use change. This issue has already been discussed in section 2.3.1.

cultivation lose soil carbon at an exponential rate (Davidson and Ackerman 1993; 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 dissertation.

Carbon stocks per unit area CS_R and CS_A include both soil and above and below ground vegetation and are calculated according to

$$CS_i = (\mathrm{SOC} + \mathrm{C_{VEG}}) \times \mathrm{A}\,, \tag{2.9}$$

in which SOC is the soil organic carbon, C_{VEG} represents the above and below ground vegetation carbon stock and A is a factor scaling to the area concerned (EC 2010a). The soil organic carbon content is given by

$$\mathrm{SOC} = \mathrm{SOC_{ST}} \times \mathrm{F_{LU}} \times \mathrm{F_{MG}} \times \mathrm{F_I}\,, \tag{2.10}$$

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.

Several authors call the amount of CO_2 emissions from land use change the “carbon debt” of land conversion (Fargione et al. 2008; Gibbs et al. 2008; Kim et al. 2009). 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].

Calculation of GHG intensity also includes emissions of nitrous oxide (N_2O) from farming activities. Because nitrous oxide has a high impact on global warming, the assessment of N_2O emissions from soil is 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).

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 can vary widely depending on soil type, climate, tillage method, fertilizer application rates and crop type (Mosier et al. 1998; Larson 2006; JEC 2007; Reijnders and Huijbregts 2008; Stephenson et al. 2008; Crutzen et al. 2008). Even though the method developed by the IPCC indicates a wide error range for direct and indirect soil N_2O emissions in an effort to cope with all those factors, this is not sufficient to cover the range of measurements available from individual fields (JEC 2007). Ideally, local N_2O emissions are measured empirically, although common methods for measuring N_2O emissions are expensive and face practical challenges (McBride et al. 2011). Continuous effort to improve the accuracy of soil emission estimates is thus important (JEC 2007).

An alternative to direct measurements is the use of models to estimate N_2O fluxes from soils (Adler et al. 2007; Bouwman et al. 2010). Different models are available that take into account a variety of factors, although the interaction between several factors is still not well understood (Farquharson and Baldock 2008). More and better data from field experiments may however lead to improved models for estimating N_2O emissions in the future (Whitaker et al. 2010). Even failing to capture site- and management-specific variations, the use of default emission factors in models that estimate N_2O emissions from N fertilizer application rates is considered an appropriate approach for calculating global emissions (Del Grosso et al. 2010).

GHG ACCOUNTING. The greenhouse gases considered in the calculation of life-cycle GHG emissions are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), with average global warming potentials (100-year time horizon) of GWP_{CO2}=1 (CO_2 is the reference), GWP_{CH4}=25, and GWP_{N2O}=298. Other GHG emissions from biofuel systems are usually found to be negligible and are not pursued (Wicke et al. 2008; Cherubini 2010). 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 practical terms, GHG emissions in each process 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.

2.4. FRAMEWORK IMPLEMENTED

This dissertation addresses the life-cycle energy renewability efficiency and GHG intensity (g CO_2eq MJ^{-1}) of first-generation biofuel systems in Europe. Each biofuel chain is modeled taking into account the energy and GHG emissions required to deliver the biofuel product to the final user, and thus includes biomass cultivation, processing, transportation and storage, followed by biofuel production, storage and distribution. Fossil diesel or gasoline are used as reference systems, depending on the specific biofuel system under analysis. Reference systems include extraction, transport and refining of crude oil, and distribution of final fuel.

The functional unit shall enable the comparison of energy and GHG balances between the alternative biofuel product systems under investigation, as well as with their fossil fuel comparators. The functional unit chosen 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 dissertation proposes a comprehensive framework to incorporate uncertainty in the life-cycle assessment of biofuels. Several sources of uncertainty are investigated, namely uncertainty related to parameters (parameter uncertainty) and uncertainty due to choices (scenario uncertainty). Following this methodology, 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 biofuel life-cycle studies instead of using average (deterministic) approaches can be evaluated.

A robust approach is used to address and incorporate parameter uncertainty in the life-cycle modeling of first-generation European biofuels. The main steps integrating 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[9]) 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.

Chapter 6 presents the results, as well as further details, concerning the application of sensitivity analysis, uncertainty propagation analysis and uncertainty importance analysis in biofuel systems in Europe. Next chapter presents a comprehensive review of biofuel life-cycle studies in Europe, including how parameter uncertainty has been taken into account, even if in a few studies only.

Concerning scenario uncertainty, chapter 4 describes how different co-product treatment approaches are taken into account in the life-cycle energy and GHG assessment of the biofuel systems addressed. These approaches include sub-division of the system, the substitution method, and several allocation approaches (based on output weight, energy content, and economic value). Moreover, a sensitivity analysis to different time horizons is conducted, aiming to evaluate the implications of alternative options that prioritize long- and short-term reductions in global warming. The most commonly used time horizon of 100-years is chosen as a baseline for GWP estimation. Taking into account the short- to mid-term implications of first generation biofuels in

[9] The Oracle Crystal Ball software package (v.11.1) was used to perform Monte-Carlo simulation (Oracle 2010).

terms of global warming effect, a sensitivity analysis to a timeframe of 20-years is also presented. Finally, results for the remaining time horizon routinely reported by the IPCC (500-years) are included for comparison purposes. An uncertainty of ±35% for the 90% confidence range is considered for GWP_{CH4} and GWP_{N2O}, according to IPCC (2007).

2.5. CONCLUDING REMARKS

This chapter shows how biofuel life-cycle studies capturing uncertainty are subject to a wide variety of influencing aspects, which include different assumptions and modeling options, lack of knowledge concerning parameters, and methodological limitations. Practical aspects of the framework implemented to address uncertainty in the life-cycle energy and GHG assessment of European biofuel systems are also presented. Next chapter presents a review of recently published life-cycle studies for biofuel systems in Europe, and demonstrates the relevance of taking into account several modeling issues in the overall energy and GHG balances of biofuels.

3. Review of Biofuel Life-Cycle Studies in Europe

"Of course, GHG balances are intimately correlated with the energy balances…"
Frondel and Peters (2007), after reviewing several biodiesel life-cycle studies.

"LCE [life-cycle energy] and LCG [life-cycle GHG] studies are closely related because fossil fuel use has a large influence on both net energy efficiency and GHG emissions"
Liska and Cassman (2008)

"Since GHG emissions are strongly linked to fossil fuel use (and N_2O emissions), the GHG emission indicator displays similar trends as the energy indicators"
de Vries et al. (2010)

* * *

"… taking into account soil emissions in biofuel LC assessments negates the correlation between nonrenewable energy inputs and GHG emissions…"
Malça and Freire (2011a)

"However, when SOC [soil organic carbon] is included in the analysis, primary energy use does not correlate with GHG emissions …, so that it cannot be used as a proxy."
Brandão et al. (2011)

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3. REVIEW OF BIOFUEL LIFE-CYCLE STUDIES IN EUROPE

3.1. PURPOSE AND SCOPE

This chapter has several goals:

- Firstly, to present a comprehensive review of life-cycle studies published in recent years (since 1998) for Rapeseed Methyl Ester (RME) in Europe. Studies are compared in terms of non-renewable primary energy requirement and GHG intensity of biodiesel. A detailed description of relevant aspects, including modeling choices, is included to identify the main causes for the high variability of results from the various biodiesel assessments;
- Secondly, to demonstrate that the correlation between nonrenewable energy inputs and GHG emissions presented by most former studies – which did not consider N_2O emissions due to land use and carbon emissions due to land use change – does not hold. Actually, a direct correlation between how soil emissions are modeled and increasing values for calculated biofuel GHG intensity has been found;

- Thirdly, to present a survey of recently published reviews on wheat- and sugar beet-based bioethanol life-cycle studies in Europe. Special emphasis is given to identifying the factors underlying the wide variability of energy requirement and GHG intensity results that has been found.

3.2. BIODIESEL STUDIES[1]

3.2.1. Methods

This section presents the main findings from a literature review conducted on life-cycle energy and GHG emissions assessment of rapeseed-based biodiesel (RME) for Europe. An online search of publicly available articles and reports has been conducted to find studies published in recent years (since 1998) with detailed information on the methodology, assumptions, and data used. A total of more than forty studies have been assessed, of which a selection of 27 is presented in Table 3.1. The remaining studies have been excluded from Table 3.1 due to lack of transparency or sufficient quantitative information.

The main results of the surveyed studies in terms of non-renewable primary energy requirement (E_{req}) and greenhouse gas (GHG) intensity of biodiesel are summarized in Fig. 3.1. The non-renewable primary energy requirement ($E_{req} = \sum E_{non\text{-}ren,prim}$ / FEC) is calculated by evaluating all the non-renewable energy inputs ($\sum E_{non\text{-}ren,prim}$) in upstream processing steps like agriculture, transportation and processing, which are compared against the biofuel final energy content (FEC), measured in terms of lower heating value (LHV). The GHG intensity (g CO_2eq MJ_f^{-1}) quantifies the amount of GHG emissions per unit of FEC. For some surveyed studies, the original outcomes have been further calculated to express the results in terms of E_{req} and GHG intensity, as defined above. Studies for which there is a range of results are represented in Fig. 3.1 by a line connecting the points that define the corresponding lower and upper limits. The correspondence between data labels used in Fig. 3.1 and surveyed studies is indicated in

[1] This section has been published in "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*, vol. 15, iss. 1, pp. 338-351".

Table 3.1. Results from studies that calculated only energy or GHG emissions are shown on the respective axis.

Surveyed study [a]	De Nocker et al. (1998)	IEA (1999) [b]	Richards (2000)	Scharmer (2001)	ADEME (2002) [c]	GM (2002)	Mortimer et al. (2003)	Bernesson et al. (2004)	JEC (2004)
Data label in Figs. 3.1, 3.2	**(98)**	**(99)**	**(00)**	**(01)**	**(02a)**	**(02b)**	**(03)**	**(04a)**	**(04b)**
Relevant data, choices and assumptions									
Geographical scope	Belgium [d]	n/d	United Kingdom	Europe	France	Europe	UK	Sweden	Europe
Temporal scale	1996 to 1998	1992 to 1996	1994 to 2000 data	1994 to 2001 data	2002; prospect. up to 2009	1995 to 1999 data	1996	1990-2001 data	2010 - 2020
System boundaries	WtW	WtT; WtW	WtT	WtT	WtT [e]	WtT [e,f]	WtT	WtT	WtT; TtW; WtW
Functional Unit	kg biodiesel	GJ biodiesel	MJ biodiesel; hectare.year	tonne biodiesel	MJ biodiesel	MJ biodiesel	tonne biodiesel	MJ biodiesel	MJ biodiesel; km traveled
Co-product credit approach	No	n/d	Substitution: straw for energy; rape meal as fertilizer; glyc for process energy	Energy allocation + Substitution: rape meal as animal feed; glyc for synthetic glyc	Mass allocation + Substitution	Substitution: rape meal as animal feed; glycerin as fuel or replacing synthetic glycerin	Economic allocation (rape straw, rape meal and glycerin)	Energy and economic allocation + Substitution: rape meal as animal feed; glyc for synthetic glycerin	Substitution: rape meal as animal feed; glycerin for animal feed or propylene glycol
Capital goods	No	No	No	Yes (n/d)	Yes (n/d)	No	1.0% (energy) 0.5% (GHG)	1.4% (energy) 0% (GHG) [l]	No
Agric. ref. system	No	n/d	No	Set-aside	n/d	Set-aside	Set-aside	No	No
Carbon emissions from land use change (dLUC)	No	n/d	No	No	Yes (n/d)	No	No	No	No
N_2O emissions from land use [kg N_2O ha^{-1} yr^{-1}]	No	n/d	Yes (1.80)	Yes (3.78)	Yes (0.5% of the N applied) [g]	Yes (4.89; 0.77 min; 13.97 max)	Yes (0.71)	Yes (2.74)	Yes (4.15; 2.91 min; 5.40 max)
Type of LCA	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional
Indirect Land Use Change	No	n/d	No	No	No	No	No	No	No
Parameter uncertainty	No	n/d	No	No	No	Yes (Monte-Carlo)	Yes (ranges with upper/lower limits)	Yes (single parameter sensitivity analysis)	Yes (Monte-Carlo)
Selected results									
Energy Requirement E_{req} [MJ_p MJ_f^{-1}]	0.524	0.40; 0.66; 0.92 (min; avg; max)	0.27 (w/ straw); 0.457 (w/o straw)	0.338 to 0.439 [h]	0.334	-0.06 to 0.40 ± 0.01	0.437 ± 0.024 (conv prod) 0.208 ± 0.017 (modified prod) [i]	-0.367; 0.355 (small scale, subst; small scale, economic)	0.44 (glycerin as feed) 0.39 (glyc for glycol)
GHG intensity [g CO_2eq MJ_f^{-1}]	46.7	No	48.2 (w/o straw); 50.5 (w/ straw)	34.4 (w/o soil N_2O); 45.9 (w/ soil N_2O)	20.2 to 23.7	10.9 to 77.7	40.6 ± 2.4 (conv prod) 18.8 ± 1.4 (modified prod)	30.9; 51.1 (large scale, subst; small scale, economic)	32.9; 53.9; 73.3 (min; default; max) (glycerin as feed) 28.8; 48.9; 68.9 (glycerin for glycol)

Table 3.1. Surveyed LC studies of biodiesel (RME) production in Europe: relevant data and assumptions, methodological choices and key results.

Surveyed study [a]	Janulis (2004)	Malça and Freire (2004c)	Dewulf et al. (2005)	SenterNovem (2005b)	Fredriksson et al. (2006)	Lechón et al. (2006)	Mortimer and Elsayed (2006)	Wagner et al. (2006)	JEC (2007)
Data label in Figs. 3.1, 3.2	**(04c)**	**(04d)**	**(05a)**	**(05b)**	**(06a)**	**(06b)**	**(06c)**	**(06d)**	**(07b)**
Relevant data, choices and assumptions									
Geographical scope	Lithuania	France	Sweden	Various [j]	Sweden	Spain	UK (North East)	Germany	Europe
Temporal scale	n/d	1993 to 2004 data	1997/99 data	2005-2008	1994-2004 data	2006	2005	1996-2002 data	2010-2020
System boundaries	WtT	WtT	WtT	WtW	WtW	WtW	WtT	WtT, WtW	WtT; TtW; WtW
Functional Unit	tonne biodiesel	MJ, liter and kg of FD; hectare.year	hectare.year	km traveled	1000 hectare.year	km traveled	tonne biodiesel	kWh; km traveled	MJ biodiesel; km traveled
Co-product credit approach	Energy allocation (straw, rape meal and glycerin)	Mass, energy and economic allocation + Substitution: rape meal as animal feed; glyc for synthetic glycerin	Exergy allocation (straw, rape meal and glycerin)	Economic allocation + Substitution: rape meal as animal feed	Economic allocation (co-products n/d)	Economic allocation + Substitution: glyc for synthetic glycerin or residue	Economic allocation + Substitution: rape meal as animal feed or biomass co-firing [k]	Energy allocation	Substitution: rape meal as animal feed; glycerin for animal feed or propylene glycol
Capital goods	8.8% (energy)	No	No	No	No	No	2.8% (energy) 2.2% (GHG)	No	No
Agric. ref. system	No	Set-aside	No	Set-aside	No	Set-aside	Set-aside	n/d	No
Carbon emissions from land use change (dLUC)	No	No	No	No	No	No	No	n/d	No
N_2O emissions from land use [kg N_2O ha^{-1} yr^{-1}]	No	No	No	Yes (2.56 to 5.60)	Yes (n/d)	Yes (0.4% of the N applied; 0.25% min; 2.25% max) [g]	Yes (4.36)	n/d	Yes (3.12 ± 1.23)
Type of LCA	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional
Indirect Land Use Change	No	No	No	No	No	No	No	n/d	No
Parameter uncertainty	No	No	No	Yes (single parameter sensitivity analysis)	Yes (single parameter sensitivity analysis)	Yes (single parameter sensitivity analysis)	Yes (ranges with upper/lower limits)	No	Yes (Monte-Carlo)
Selected results									
Energy Requirement E_{req} [MJ_p MJ_f^{-1}]	0.193 to 0.446	0.335 to 0.41	0.324 (exergy basis)	0.49 + 10% 0.49 – 20%	0.132 + 14.4% 0.132 – 9.6% [l]	0.184; 0.259; 0.747 [m]	0.54 ± 0.026 (animal feed) 0.041 ± 0.03 (biomass co-firing)	0.37	0.51 (glycerin as feed) 0.46 (glyc for glycol)
GHG intensity [g CO_2eq MJ_f^{-1}]	No	13.0 to 23.0	No	50.3 ± 40%	22.1 + 26.2% 22.1 – 13.4% [l]	29.6; 37.6; 60.3	53.8 ± 2.2 (animal feed) 37.5 ± 2.8 (biomass co-firing)	24.9	30.7; 51.8; 68.3 (min; default; max) (glycerin as feed) 25.3; 46.5; 66.6 (glyc for glycol)

Table 3.1 (cont). Surveyed LC studies of biodiesel (RME) production in Europe: relevant data and assumptions, methodological choices and key results.

Surveyed study [a]	Hansson et al. (2007)	Harding et al. (2007)	Zah et al. (2007)	Halleux et al. (2008)	Reijnders and Huijbregts (2008)	Stephenson et al. (2008)	Lechón et al. (2009)	Soimakallio et al. (2009)	Thamsiriroj et al. (2009)
Data label in Figs. 3.1, 3.2	**(07c)**	**(07d)**	**(07e)**	**(08a)**	**(08b)**	**(08c)**	**(09a)**	**(09b)**	**(09c)**
Relevant data, choices and assumptions									
Geographical scope	Sweden	n/a	Switzerland [n]	Belgium	Europe	United Kingdom	Spain	Finland	Ireland
Temporal scale	1994 to 2004 data	2002/03 data	2004	2005	2002 to 2007 data	2006/07 data	2008; prospective up to 2020	2003/06 data	2003/06 data
System boundaries	WtW	WtT	WtW	WtW	WtT	WtT	WtW	WtW	WtT [e]
Functional Unit	1000 hectare.year	tonne biodiesel	MJ biodiesel; hectare.year; person.km	100 km traveled	MJ biodiesel; kg biodiesel	tonne biodiesel	km traveled	km traveled	hectare.year
Co-product credit approach	Economic allocation (co-products n/d)	Mass allocation (glycerin)	Economic allocation	Substitution: rape meal as animal feed; glycerin for chemicals	Economic allocation (rape meal) (glycerin n/d)	Economic allocation + Substitution: rape meal and glycerin for CHP co-firing	Substitution: rape meal as animal feed; glyc for synthetic glycerin, propylene glycol or residue	Substitution: rape meal as animal feed; glycerin for heat in boilers	No
Capital goods	No	No	21% – 27% (GHG) [o]	No	No	4.7% – 5.9% (energy) 0.9% – 1.2% (GHG)	No	No	No
Agric. ref. system	No	n/a	No	No	No	Set-aside	Set-aside	Set-aside	No
Carbon emissions from LUC (dLUC) [t CO_2 ha^{-1} yr^{-1}]	No	n/d	No	No	Yes (3.08)	No	No	Yes (-0.011 to 0.286) [p]	No
N_2O emissions from land use [kg N_2O ha^{-1} yr^{-1}]	Yes (n/d)	n/d	Yes (1.6% to 3.5% of the N applied) [g]	Yes (n/d)	Yes (2.45 to 8.20)	Yes (1.60 small scale; 2.11 large scale)	Yes (0.4% of the N applied) [g]	Yes (2.55; 0.40 min; 11.18 max)	Yes (1.70)
Type of LCA	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional	Attributional
Indirect Land Use Change	No	No	No	No	No	No	No	No	No
Parameter uncertainty	No	No	Yes [q]	No	Partially (for N_2O emissions)	No	No	Yes (Monte-Carlo)	No
Selected results									
Energy Requirement E_{req} [MJ_p MJ_f^{-1}]	0.120	No	0.68	0.132	0.60 [r]	0.538 (small scale) 0.552 (large scale)	0.21 (imported rape) 0.33 (domestic rape)	0.5 ± 0.15	0.456
GHG intensity [g CO_2eq MJ_f^{-1}]	21.8	107.3 to 117.5	50.7; 67.2; 89.5 (min; avg; max)	15.1	123.7 to 147.8	58.8 (small scale) 64.7 (large scale)	35.4 (imported rape) 76.2 (domestic rape)	80; 100; 170	62.2

Table 3.1 (cont). Surveyed LC studies of biodiesel (RME) production in Europe: relevant data and assumptions, methodological choices and key results.

Footnotes to Table 3.1: [a] Each surveyed study is labeled for identification purposes in Figs. 3.1 and 3.2; [b] Several data is not distinguishable because IEA (1999) is a review of different studies; [c] Only the executive summary was available in the web; therefore, even though a sensitivity analysis has been performed for emissions from cultivated soil, a detailed analysis of the implications of this analysis could not be made; [d] It also included Western European data, when specific data for Belgium was not available; [e] Well-to-tank study plus theoretical calculation of combustion on the basis of the carbon content of the fuels; [f] WtW assessments for different combinations of (bio)fuels and powertrains, but not for rapeseed-based biodiesel, were assumed; [g] Fertilizer application rates not distinguishable in order to calculate soil N_2O emissions from land use in kg N_2O ha^{-1} yr^{-1}; [h] This study assumes different cultivation locations and different scales for industrial conversion; [i] Modified, as opposed to conventional, production of biodiesel from oilseed rape consists of low-nitrogen cultivation of oilseed rape, the use of rape straw as an alternative heating fuel in the processing of biodiesel, and the replacement of conventional diesel by biodiesel in agricultural operations and road transport vehicles; [j] UK, The Netherlands, Germany, France and Poland; [k] Co-product glycerin was only dealt with by means of economic allocation; [l] A sensitivity analysis has been conducted to evaluate the effect of ± 20% changes in input data, e.g. crop yield, tractor and soil emissions, oil extraction efficiency and oil price; [m] In addition to different co-product approaches, sensitivity of results has been tested to the origin of rapeseed and the energy efficiency of the industrial conversion stage; [n] The study covers Swiss and foreign renewable energy production, but only Switzerland for the consumption of renewable energy; [o] Includes the production and maintenance of vehicles and construction and maintenance of roads; [p] Negative value means carbon sequestration; [q] Covers only the uncertainty in the gathering of inventory data; [r] Cumulative energy demand data (MJ MJ^{-1}) from Zah et al. (2007) was used; n/a: not applicable; n/d: not distinguishable.

3.2.2. Comprehensive analysis of surveyed studies

BIODIESEL NON-RENEWABLE PRIMARY ENERGY REQUIREMENT AND GHG EMISSIONS. Non-renewable primary energy requirement of biodiesel (E_{req}) for the surveyed studies is shown in Fig. 3.1. Fossil diesel (FD) is also represented and used as a baseline reference. It can be observed that biodiesel E_{req} results present significant variations, ranging from 0.92 MJ_p MJ_f^{-1} (the highest value, presented in a review performed by IEA 1999) to negative energy requirements (GM 2002; Bernesson et al. 2004). Negative values can be calculated when energy credits greater than the energy inputs are given to the biodiesel chain: e.g. GM (2002) and Bernesson et al. (2004), which assume that the co-product glycerin from transesterification displaces the energy intensive production of synthetic glycerin. The E_{req} results for the majority of studies fall in the range of 0.15 to 0.60 MJ_p MJ_f^{-1}. This is a broad range, but clearly below the fossil diesel E_{req}, meaning that net reductions in fossil energy consumption are obtained when biodiesel displaces fossil diesel. The large variations in biodiesel E_{req} in deterministic studies can be explained by the adoption of different approaches for the treatment of co-products and different assumptions in the agricultural and industrial stages (Richards 2000; Mortimer et al. 2003; Janulis 2004; Lechón et al. 2006; Lechón et al. 2009). A few studies include parameter uncertainty, which results in large variations in E_{req} (GM 2002; JEC 2007; Soimakallio et al. 2009).

Regarding the life-cycle GHG emissions of biodiesel, Fig. 3.1 shows a very high range of emissions for the surveyed studies, with results ranging from 15 to 170 g CO_2eq MJ_f^{-1}. This range is broader than the one observed for E_{req} results, particularly when fossil diesel results are taken as a reference. In general, recent studies present higher values – above 60 g CO_2eq MJ_f^{-1}–, which are near or even above fossil diesel GHG intensity. A few recent studies, in particular, show a very high GHG intensity (above 100 g CO_2eq MJ_f^{-1}) for biodiesel (Reijnders and Huijbregts 2008; Soimakallio et al. 2009), which is explained by a very high contribution from carbon and N_2O emissions from soil. Nonetheless, several recent studies also indicate quite low GHG emissions for biodiesel (Fredriksson et al. 2006; Hansson et al. 2007; JEC 2007; Halleux et al. 2008).

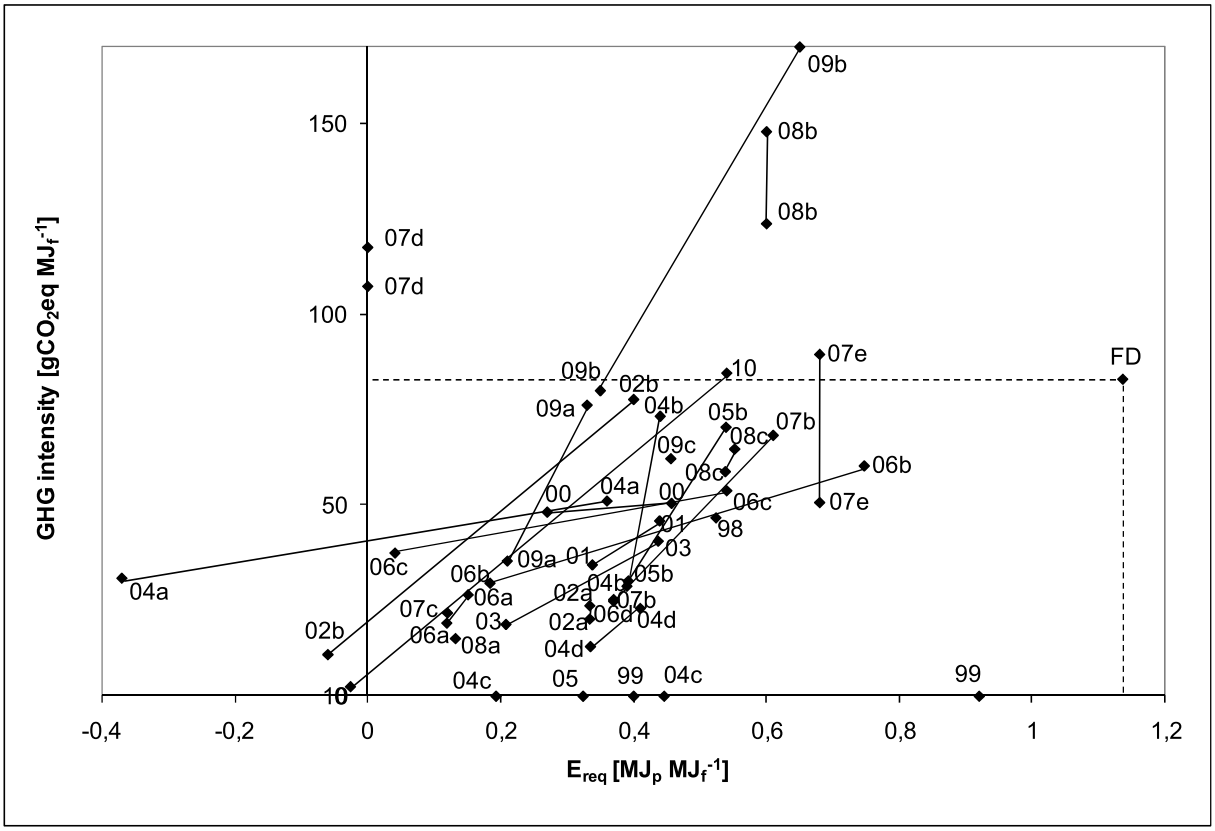


Fig. 3.1. GHG intensity and non-renewable primary energy requirement E_{req} for biodiesel life-cycle studies in Europe (data labels represent surveyed studies as indicated in Table 3.1; FD: fossil diesel).

To facilitate calculation of non-renewable energy savings and avoided GHG emissions when biodiesel displaces fossil diesel (FD), the latter is also represented in Fig. 3.1, with 1.14 $MJ_p\ MJ_f^{-1}$ and 82 g CO_2eq MJ_f^{-1} (average values presented by Hekkert et al. 2005, on the basis of data from 15 studies). Biodiesel studies within the area delimited by the dashed lines have both lower GHG intensity and lower E_{req} than fossil diesel.

Results from most former studies report a correlation between biodiesel non-renewable energy inputs and GHG emissions, as emphasized in the review by Frondel and Peters (2007). However, the results presented in Fig. 3.1 do not show a general mathematical relationship between GHG intensity and non-renewable energy requirement. The importance of soil emissions in terms of the overall GHG intensity means that taking into

account soil emissions in biofuel life-cycle assessments negates the correlation between non-renewable energy inputs and GHG emissions presented by most former studies, which did not consider N_2O emissions due to land use and carbon emissions due to land use change (LUC).

The broad range of E_{req} and GHG intensities presented in Fig. 3.1 stresses the need to understand the main drivers for the differences and variations between different studies (and also within specific studies): are they due to different methodological procedures (or modeling choices), data or production conditions? A comprehensive discussion on the key issues that may affect the life-cycle performance of biofuels follows. These include: geographical scope and system boundaries; functional unit; assessment of co-products; energy and emissions associated with facilities and machinery; reference land use; soil emissions due to land use and land use change; type of LCA approach; and parameter uncertainty. Relevant data from each surveyed study, including major assumptions, methodological choices and results, are gathered in Table 3.1. Studies are listed in chronological order.

GEOGRAPHICAL SCOPE AND SYSTEM BOUNDARIES. The majority of reviewed studies focus on specific European countries, and seven are European-wide assessments. Depending on the study, relevant data for the main stages in biodiesel life-cycle (cultivation of raw materials and industrial conversion) spans from a few years (two or less) to over a decade.

Concerning the system boundaries considered in the studies reviewed in Table 3.1, different life-cycle approaches were adopted. The majority of studies (19 out of 27) adopted a well-to-tank approach. About 13 studies adopted a full well-to-wheels (WtW) assessment. The “well-to-wheels” modeling boundary includes both the “well-to-tank” (WtT) and “tank-to-wheels” (TtW) stages. An example is the JEC (2007) detailed report which splits the analysis in the WtT and TtW counterparts and finally aggregates the results in a full WtW assessment.

The majority of WtW studies assessed the TtW stage using the fuel consumption of a typical passenger vehicle or performing theoretical combustion calculations. In the

studies performed by the JEC consortium (JEC 2004, 2007), a vehicle simulation tool developed by NREL was used to simulate a compact sized 5-seater European sedan, which enabled the comparison of different (bio)fuels and associated powertrains. Simulation figures were cross-checked with experimental data from a specific top selling model of a European car manufacturer. Lechón et al. (2006, 2009) evaluated the use of different fuels and fuel mixes in a specific vehicle model, which was selected as representative of the Spanish passenger car fleet. The new European driving cycle as defined in EPC (1998) was adopted in the study. Tailpipe CO_2 emissions were calculated on the basis of the carbon content of fuels. Other GHG emissions were estimated from literature data and equal emissions were assumed for both biodiesel and fossil diesel combustion. Wagner et al. (2006) compared different fuels and propulsion concepts in a medium size automobile operated in the new European driving cycle. The energy efficiency of internal combustion engines running on biofuels and fossil fuels was evaluated on the basis of data from several car manufacturers. SenterNovem (2005b) combined the passenger car composition for the Netherlands with the emission limits for Euro 1-4 specifications (EPC 1998; EC 1999) to estimate average emissions for the Dutch car park. Vehicular emissions from the use of biodiesel and fossil diesel were estimated from Van Walwijk et al. (1999). A few other studies in this review also complemented the WtT approach with a theoretical calculation of combustion on the basis of the carbon content of (bio)fuels (GM 2002; ADEME 2002; Thamsiriroj and Murphy 2009).

FUNCTIONAL UNIT. The definition of the functional unit in biodiesel life-cycle studies is related with the scope and system boundaries of the study; therefore, there is no single or preferred functional unit among reviewed studies. For example, nine studies use 1MJ or 1GJ of fuel energy content (measured in terms of the lower heating value), as this is an appropriate basis for comparison of the energy delivered by a biofuel to the end user. Other studies (7 out of 27) adopt a measure of agricultural surface area (usually the hectare), emphasizing the importance of land use impacts and the scarcity problem of available land for growing energy crops. WtW approaches often use distance traveled (usually the kilometer) as the functional unit (JEC 2004, 2007; SenterNovem 2005b; Lechón et al. 2006, 2009; Wagner et al 2006; Halleux et al. 2008; Soimakallio et al. 2009).

A few studies use more than one functional unit, which is motivated by different system boundaries or the application of a sensitivity analysis (Richards 2000; JEC 2004, 2007; Malça and Freire 2004c; Wagner et al. 2006; Zah et al. 2007). As discussed in chapter 2, different system boundaries may be recommended depending on the scope of the study, which may also favor the choice of different functional units.

Comparison between life-cycle studies may be difficult due to differences in the functional units adopted (van der Voet et al. 2010). In these cases, the specific results from each study have been converted to a common functional unit (1 MJ, LHV), based on the specific data included in each reviewed study, so that the outcomes presented in Table 3.1 are comparable.

MULTIFUNCTIONALITY AND ASSESSMENT OF CO-PRODUCTS. The biodiesel chain is usually multifunctional (i.e. produces more than one product). The studies reviewed used different methods, based on allocation or substitution, to handle multifunctionality. About 18 studies used allocation approaches, on the basis of underlying relationships, to partition the input and output flows of the biodiesel chain between biodiesel and its co-products. The substitution method was used in 16 studies, with various alternative scenarios being adopted. Due to the lack of a common allocation approach among studies a clear trend cannot be identified in the results presented in Fig. 3.1. Nine studies used both allocation and substitution to handle co-products and 3 studies did not use any method.

The majority of studies (12 out of 27 studies, Bernesson et al. 2004; Mortimer et al. 2003; Malça and Freire 2004c; SenterNovem 2005; Fredriksson et al. 2006; Lechón et al. 2006; Mortimer and Elsayed 2006; Hansson et al. 2007; Zah et al. 2007; Reijnders and Huijbregts 2008; Stephenson et al. 2008; Malça and Freire 2011a) used economic allocation, in which co-products are allocated according to their market prices. This method is very practical, since it uses the economic value as the main driver (Guinée et al. 2004). Nevertheless, the volatility of market prices is pointed out as the main drawback of this method, as it may strongly influence the results of the life-cycle study. Other authors prefer relatively fixed physical relationships between co-products, rather than varying economic prices, namely energy (Scharmer 2001; Janulis 2004; Wagner et

al. 2006), mass (ADEME 2002; Harding et al. 2007), and exergy (Dewulf et al. 2005). According to ISO 14044:2006, whenever several allocation approaches seem applicable, a sensitivity analysis shall be conducted to illustrate how different methods change the results. However, only Bernesson et al. (2004) and Malça and Freire (2004c) used more than one allocation approach, in order to evaluate the implications of choosing different allocation methods. These authors concluded that the results were largely dependent on the method chosen for allocation of the environmental burdens between biodiesel and its co-products.

Sixteen studies in this survey used the substitution method and expanded the biofuel system to include alternative functions for co-products, which are then regarded as credits to the chain. These alternative applications can be diverse, as detailed in Table 3.1: rape meal is used as fertilizer, animal feed, and in co-firing, whereas glycerin is used for process energy, animal feed, and displacing propylene glycol or synthetic glycerin. Various studies, in particular, assessed co-products only through substitution (Richards 2000; GM 2002; JEC 2004, 2007; Halleux et al. 2008; Lechón et al. 2009; Soimakallio et al. 2009). According to JEC (2007) and Weidema and Schmidt (2010), the substitution approach should be in most cases the preferred method, because it attempts to model reality by tracking the likely fate of co-products. It is therefore important that realistic, as opposed to academic, substitution alternatives are chosen when this method is adopted.

ENERGY AND EMISSIONS ASSOCIATED WITH FACILITIES AND MACHINERY. A few studies considered the energy and emissions associated with the construction and maintenance of capital goods. Energy embodied in agro-machinery, vehicles and processing plants represents between 1.4% and 8.8% of the total energy requirement for biodiesel production, whereas GHG emissions amount to 0.9% to 2.2% of the life-cycle GHG intensity of biodiesel (Bernesson et al. 2004; Janulis 2004; Mortimer and Elsayed 2006; Stephenson et al. 2008). The exception is Zah et al. (2007), to which GHG emissions of capital goods represent 21% to 27% of the life-cycle GHG intensity. This may be explained by the inclusion of a road maintenance stage in the inventory phase. The majority of studies, however, neglected capital goods, acknowledging that they

represent only a small fraction of the entire energy and GHG balances. The same approach is followed in this dissertation, as will be discussed in chapter 4.

REFERENCE LAND USE. Several studies (10 out of 27) considered a reference agricultural system consisting of set-aside land to which the rapeseed cultivation system is compared. This hypothesis was in line with European Common Agricultural Policy (CAP) rules in force until 2008, in which set-aside obligations were imposed –farmers were required to leave 10% of their land on set-aside–, allowing, however, the cultivation of energy crops on set-aside areas. These obligations, along with a special aid for energy crops of 45€ ha^{-1} introduced by the 2003 CAP reform, created a favorable environment for the cultivation of energy crops (Wiesenthal et al. 2009). The set-aside policy changed in 2008, when EU agriculture ministers reached a political agreement on the abolition of compulsory set-aside from 2009 onwards (CEU 2009), which allowed farmers to maximize their production potential. Primary energy inputs and GHG emissions due to occasional mowing of set-aside areas were taken into account as credits in the biofuel life-cycle studies, since these energy inputs and related emissions would not subsist if the energy crops were cultivated in those areas instead. Three studies (IEA 1999; ADEME 2002; Wagner et al. 2006) in Table 3.1 do not indicate if a reference system was taken into account, whereas 13 studies simply did not consider any reference agricultural system, mainly because rapeseed cultivation was assumed to be within a crop rotation scheme.

CARBON EMISSIONS DUE TO LAND USE CHANGE. Of the twenty seven reviewed studies, only four considered the contribution of soil carbon emissions for the GHG balance. Soimakallio et al. (2009) used IPCC (2006) data for calculating the annual change in soil carbon balance during 100 years; upper and lower limits were considered for conventional tillage and no-tillage cultivation of rapeseed, respectively. Vleeshouwers and Verhagen (2002) developed a model to calculate carbon fluxes from agricultural soils in Europe, which includes the effects of crop, climate and soil on the carbon budget of agricultural land. According to these authors, European arable soils are estimated to lose 0.84 ± 0.40 t C ha^{-1} yr^{-1} (Vleeshouwers and Verhagen 2002; Freibauer et al. 2004). In the work by Reijnders and Huijbregts (2008), which is included in this review, the

average value of 0.84 t C ha^{-1} yr^{-1} (3.08 t CO_2 ha^{-1} yr^{-1}) was used for rapeseed cultivation in a crop rotation system.

Vleeshouwers and Verhagen (2002), Reijnders and Huijbregts (2008) and Soimakallio et al. (2009) concluded that direct soil carbon emissions from land use change is an important aspect for the GHG balance of biofuels. However, this review shows that this issue has not captured enough attention, even in recent biofuel life-cycle studies. It must be also emphasized that carbon emissions due to land use change are intimately correlated with the reference land use considered, as demonstrated e.g. by Hoefnagels et al. (2010) and Malça and Freire (2009c). It depends on the type of reference land whether the carbon content of actual land use is higher or lower than the reference situation (Hoefnagels et al. 2010).

N_2O EMISSIONS DUE TO LAND USE. Even though N_2O emissions from soil were taken into account in the majority of reviewed studies (21 out of 27), this assessment was in most cases performed with deterministic data. Several authors estimated nitrous oxide emissions using single figures for the N_2O emitted, which was calculated as a percentage of the N fertilizer input to cultivated soil (Richards 2000; Scharmer 2001; ADEME 2002; Mortimer et al. 2003; Bernesson et al. 2004; Mortimer and Elsayed 2006; Stephenson et al. 2008; Lechón et al. 2009; Thamsiriroj and Murphy 2009). A wide range of deterministic values was used across studies (see details in Table 3.1).

However, nitrous oxide emissions from soil vary widely and depend upon a number of factors, as discussed in chapter 2. A few studies in this review included uncertainty data concerning N_2O emissions, whether through ranges with upper and lower limits (SenterNovem 2005b; Lechón et al. 2006), or by using stochastic methods (GM 2002; JEC 2004, 2007; Soimakallio et al. 2009). The uncertainty ranges used are listed in Table 3.1.

TYPE OF LCA APPROACH. All surveyed life-cycle studies in Table 3.1 are attributional. More recently, a few consequential LCA studies for biodiesel in Europe have been conducted. A recently published example is the work by Reinhard and Zah (2011), who studied the environmental consequences of displacing fossil diesel imports into Switzerland by increasing domestic RME production. Due to constraints in the

availability of arable land, increased rapeseed production for energy purposes in Switzerland is achieved at the expenses of barley production and edible rapeseed oil production, which in turn are compensated for by imports from foreign countries[2]. Reinhard and Zah (2011) concluded that the environmental impacts of an increased RME production in Switzerland rather depend on the environmental scores of the marginal replacement products on the world market, than on local production factors.

As discussed in chapter 2, an aspect that requires a consequential approach is the assessment of indirect land use change[3] (iLUC). With the exception of two studies in which the inclusion of iLUC is not distinguishable, all the remaining studies do not address this issue.

PARAMETER UNCERTAINTY. Concerning the inclusion of parameter uncertainty in surveyed studies, it can be seen that former studies did not consider this type of uncertainty (De Nocker et al. 1998; Richards 2000; Scharmer 2001; ADEME 2002; Janulis 2004; Malça and Freire 2004c; Dewulf et al. 2005; Wagner et al. 2006; Hansson et al. 2007; Harding et al. 2007) or, at least, it was only considered in a simplified way through single parameter sensitivity analysis (Bernesson et al. 2004; Mortimer et al. 2003; SenterNovem 2005; Fredriksson et al. 2006; Lechón et al. 2006; Mortimer and Elsayed 2006). The exceptions are GM (2002) and JEC (2004), in which parameter uncertainty was evaluated using Monte-Carlo simulation, a technique that proves difficult in becoming standard, as recent studies that still do not include parameter uncertainty have show (Halleux et al. 2008; Stephenson et al. 2008; Lechón et al. 2009; Thamsiriroj and Murphy 2009). The adoption of probabilistic approaches to address previously neglected issues, namely soil emissions with high uncertainty (Soimakallio et al. 2009), leads to higher GHG emissions and wider biodiesel GHG ranges. Moreover, the conclusion of some former studies indicating that the results of biofuel life-cycle studies were largely dependent on the allocation method selected for co-product evaluation can be questioned when parameter uncertainty is included in the assessment, as recently

[2] Increased production in foreign countries is met either by land expansion or agricultural intensification.
[3] An exception is the work by Overmars et al. (2011) who use an attributional approach to deal with the iLUC issue. These authors explicitly calculate iLUC emissions related to EU biofuel consumption based on historical (2007) data, instead of using forward looking economic and environmental modeling approaches.

demonstrated by Malça and Freire (2010a) for the production of vegetable oil from rapeseed in Europe.

3.2.3. Prior review studies

Most former studies presented clear advantages in terms of GHG intensity for biodiesel over fossil diesel because they neglected carbon emissions from soils and were based on deterministic life-cycle models. This is the case with all surveyed studies up to 2006 in this review, with the exception of GM (2002) and JEC (2004). Other studies in the literature point out the same conclusion. For example, the International Energy Agency conducted a review of several studies, dated from 1993 to 2002, on the energy requirement and well-to-wheels GHG emission impacts from using rapeseed-derived biodiesel rather than conventional diesel fuel (IEA 2004). Main findings from this survey were that fossil energy requirement of biodiesel production systems vary between 0.33 and 0.57 MJ per MJ of biofuel energy content. The estimates for net GHG emission reductions in light-duty compression-ignition engines range from 44% to 66%. Richards (2000) also concluded that biodiesel production was strongly energy positive and, where straw was burned as fuel and oil seed rape meal used as a fertilizer, the balance was even better.

Larson (2006) conducted a review of several life-cycle studies covering a variety of conventional and future generation liquid biofuels for transportation, in which different aspects are highlighted that justify the wide range of results between studies. Due to the broadening scope of the study, only a few studies addressing rapeseed-based biodiesel were assessed in the review; for these, the main finding was that RME shows GHG emission savings compared to conventional diesel fuel.

Frondel and Peters (2007) also found that the energy and GHG balances of supporting rapeseed-based biodiesel as a substitute for fossil diesel were clearly positive. Based on a survey of empirical studies, these authors concluded that between 55% and 79% of fossil resources can be saved with the substitution. Moreover, Frondel and Peters (2007)

found that GHG balances were intimately correlated with energy balances, with estimates of GHG savings in the range of 41 to 78%.

Recently, Yan and Crookes (2009) have published a review of nine studies addressing the life-cycle of rapeseed-derived biodiesel. Depending on the study, these authors concluded that the fossil fuel use and GHG emissions for biodiesel were in the range of, respectively, 0.33-0.65 MJ_p MJ_f^{-1} and 20-53 g CO_2eq MJ_f^{-1}. This low level of emissions may be explained by deterministic assessments not accounting for N_2O or carbon emissions from soil.

Hoefnagels et al. (2010) reviewed the impact of different assumptions and methodological choices on the life-cycle GHG emissions of various biofuels (bioethanol, biodiesel and Fischer-Tropsch diesel). Key factors affecting the performance of biofuels included allocation of co-products, location of crop cultivation production, crop yields, reference land (LUC) and soil N_2O emissions. Concerning rapeseed-based biodiesel (RME), only one study was reviewed. The main conclusion is that RME GHG emissions can vary between 17 and 140 g CO_2eq MJ_f^{-1} depending on the key parameters and methodological choices considered.

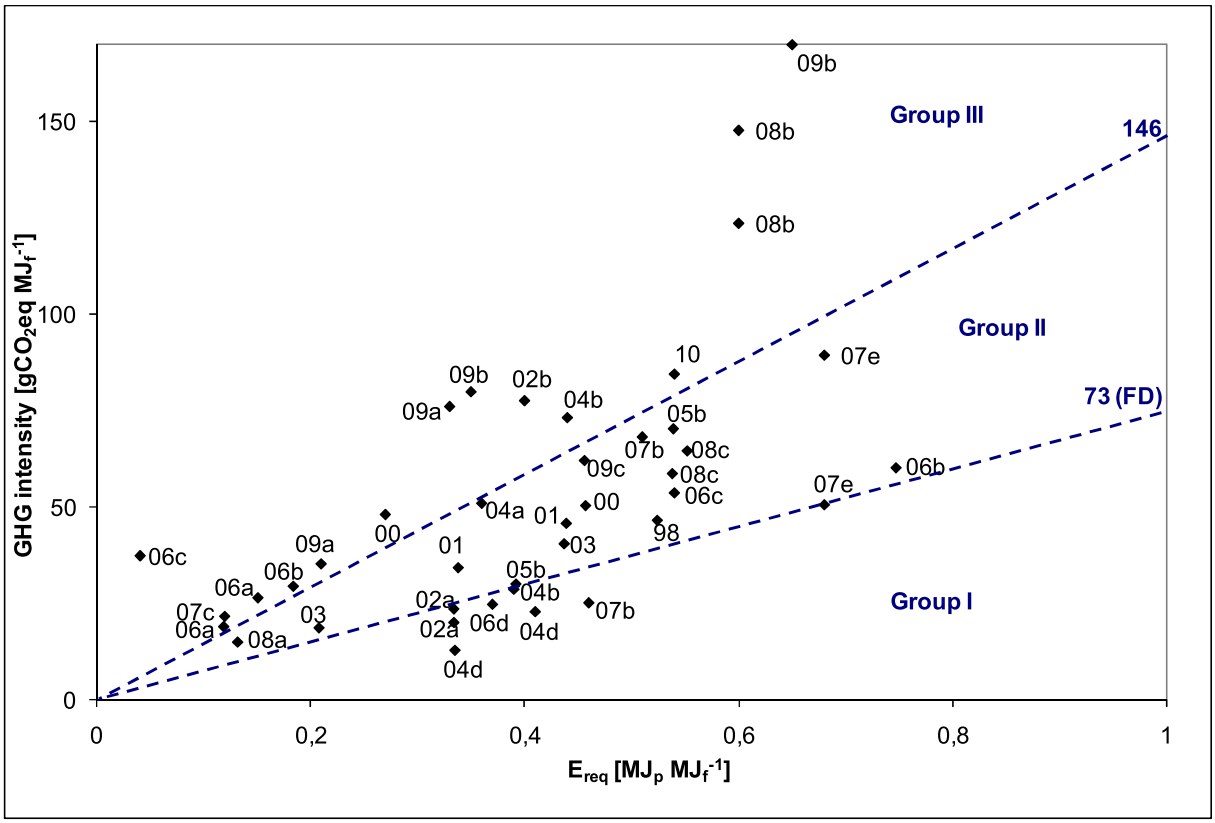
Concerning carbon emissions from LUC, Majer et al. (2009) conducted a review of biodiesel life-cycle studies and showed the significant influence of LUC effects on the potential GHG emission savings associated with biodiesel from palm and jatropha. None of the rapeseed-based biodiesel studies in the revision by Majer et al. (2009) addressed land use change issues, however.

3.2.4. Discussion

Fig. 3.2 groups the surveyed studies according to the extent to which some of the key methodological GHG issues have been addressed in the reviewed assessments, namely inclusion of N_2O and carbon emissions from cultivated soil. All reviewed studies take into account fossil CO_2 emissions throughout the life-cycle, but do not follow the same methodology concerning soil emissions. Fig. 3.2 shows that a direct linkage exists between taking into account soil emissions in biofuel life-cycle studies and increasing

values for calculated GHG emissions. Studies have been divided into three groups: Group I gathers studies that do not account for N_2O emissions from soil or at most adopt low (and deterministic) values for these emissions. Group II includes studies that account for higher N_2O emissions from soil. Group III addresses the studies that include the additional assessment of soil carbon emissions, in addition to higher nitrous oxide emissions with important uncertainty ranges.

As shown in Fig. 3.2, the classification in three groups can also be made in terms of the GHG intensity per non-renewable energy use requirement. Dashed lines in Fig. 3.2 indicate the thresholds considered for grouping: group I – studies with values below that of fossil diesel (73 g CO_2eq MJ_p^{-1}, Hekkert et al. 2005); group II – values between 73 and 146 g CO_2eq MJ_p^{-1}; and group III – values above 146 g CO_2eq MJ_p^{-1} (twice the value of fossil diesel). These results show that in Group I studies there is a correlation between biodiesel non-renewable energy inputs and GHG emissions, close to the fossil diesel value of 73 g CO_2eq MJ_p^{-1}. This means that GHG emissions are mainly due to fossil energy use. Values lower that 73 g CO_2eq MJ_p^{-1} may indicate that the mix of non-renewable energy used is less GHG intensive than fossil diesel, has it happens e.g. for biodiesel produced in France (nuclear energy) (Malça and Freire 2004c). It should be noted that Group I mainly includes former studies (up to 2006).



Group	Key modeling issues addressed	GHG intensity per non-renewable primary energy use requirement ($gCO_2eq\ MJ_p^{-1}$)		
		< 73	73 – 146	> 146
III	soil carbon emissions + soil N_2O emissions (↑, high uncert.) + fossil CO_2 emissions			10, 09b, 08b
II	soil N_2O emissions (↓ or ↑) + fossil CO_2 emissions		09c, 08c, 08a, 07e, 07b, 06c, 06b, 05b, 04b, 04a, 03, 01, 00	09a, 07c, 06c, 06a, 02b
I	soil N_2O emissions (0 or ↓) + fossil CO_2 emissions	06d, 04d, 02a	98	

Fig. 3.2. GHG intensity and non-renewable energy use requirement of surveyed studies: classification in groups; FD: fossil diesel. Data labels are defined in Table 3.1.

Groups 2 and 3 report in general more recent assessments in which further key methodological issues concerning GHG emissions not related with energy use are addressed, namely N_2O and carbon emissions from cultivated soil. In these studies, the GHG emissions per non-renewable MJ_p are considerably higher than those for fossil fuels, since GHG emissions are not exclusively linked to energy use. Soil emissions take

the lead over energy use in terms of the critical factor for overall GHG emissions. This is particularly notorious in Group 3, for which GHG emissions per non-renewable MJ_p more than double those from fossil energy use.

Recently published studies negate the definite and deterministic advantages for biodiesel presented in former studies. The reason is twofold: recent studies include soil emissions (mainly N_2O and, not as often, carbon associated with LUC) and take into account uncertainty related to parameters. In this review, Soimakallio et al. (2009) and Reijnders and Huijbregts (2008) present very high biodiesel GHG emissions, much higher than for the other assessed studies, which is due to high GHG emissions from soil with significant uncertainty. Even though direct carbon emissions from land use change may strongly contribute for the GHG balance of biofuels, this review shows that this issue has not been commonly addressed, even in recent biofuel life-cycle studies.

Another important conclusion from this review negates the correlation between biodiesel non-renewable energy inputs and GHG emissions reported in most former studies. Results presented in Fig. 3.2 do not show a general mathematical relationship between GHG intensity and non-renewable primary energy requirement. The importance of soil emissions in terms of the overall GHG intensity means that taking into account soil emissions in biofuel life-cycle assessments negates the correlation between non-renewable energy inputs and GHG emissions presented by most former studies, which did not consider N_2O emissions due to land use and carbon emissions due to LUC. Therefore energy cannot be used as a proxy for emissions, as also shown in Reijnders and Huijbregts (2008), Soimakallio et al. (2009) and Brandão et al. (2011).

This review shows how different key issues in life-cycle studies of biodiesel affect the outcomes in terms of primary energy consumption and greenhouse gas emissions. It is demonstrated that taking account of parameter uncertainty for certain key inputs (e.g. N_2O and carbon emissions from soil), as well as selection of different options for dealing with co-products (scenario uncertainty), has a strong influence in the results.

3.3. BIOETHANOL STUDIES

A review concerning wheat- and sugar beet-based bioethanol production systems in Europe is presented. Table 3.2 summarizes the main findings from the review, including E_{req} and GHG intensity results. Recently, several review studies have been published in which the energy and GHG balances of (first-generation) bioethanol production systems are addressed from a life-cycle perspective (Menichetti and Otto 2009; de Vries et al. 2010; Hoefnagels et al. 2010; van der Voet et al. 2010; Whitaker et al. 2010). This section analyzes five published works which are themselves review assessments of single bioethanol life-cycle studies, as opposed to the assessment of biodiesel systems (section 3.2) which draws on a systematic literature review conducted on several (twenty-seven) single life-cycle studies. Table 3.3 lists all the (wheat and sugar beet) bioethanol studies addressed by the review works indicated in Table 3.2. Altogether, the five review works addressed 29 wheat studies and 24 sugar beet studies.

Menichetti and Otto (2009) conducted a review of eight life-cycle studies covering wheat- and/or sugar beet-based bioethanol. These authors found a wide range of results concerning energy requirement and GHG intensity, which can be explained by the different methodological and data assumptions required to perform the life-cycle studies. Menichetti and Otto (2009) also identified the main parameters and life-cycle stages affecting the results. The agricultural phase is responsible for a significant share of GHG emissions –nitrous oxide emissions are particularly relevant due to high rate of fertilizer application–, whereas the impacts of energy use are significant in the technology conversion phase. Despite the range of results, all studies converge on indicating that the use of wheat- and sugar beet-based bioethanol leads to net benefits (in energy and GHG emissions) compared to gasoline. The GHG assessment does not include GHG emissions associated with land use change, which also explains the GHG savings reported over gasoline for all reviewed studies.

De Vries et al. (2010) examined a total of twelve published studies concerning bioethanol from wheat (9 studies) and sugar beet (9 studies). Results are very similar for the two chains, either in terms of energy requirement or GHG intensity ranges. Key determinants of results (and their ranges) are the energy inputs and GHG emissions

considered throughout life-cycle stages, and different allocation methods. Positive energy and GHG savings over gasoline are reported for both chains, which in the latter is partially explained by the limitation of GHG assessment to scenarios without land use change.

Hoefnagels et al. (2010) examined three life-cycle studies covering bioethanol production from wheat and sugar beet. These authors aimed at showing the impact of underlying assumptions and methodological choices, namely soil N_2O emissions, dLUC emissions and co-product treatment approaches, in the biofuel energy and GHG performance. Despite the large variation of results, both chains show advantages over gasoline in terms of non-renewable primary energy consumption. The same conclusion is not valid however for GHG emissions, as the results for wheat demonstrate. Taking into account carbon emissions from direct land use change (dLUC) increases significantly the uncertainty range of results. The very different LUC scenarios considered may result in negative GHG emissions (i.e. the biofuel chain is a GHG saver) or, conversely, in very high (and above gasoline) GHG emissions.

Van der Voet et al. (2010) examined the causes for variability of results among 67 life-cycle studies of biofuels, including 6 studies for wheat bioethanol and 5 for sugar beet bioethanol. A central determinant in the variability of life-cycle results is the co-product treatment approach used. The authors concluded that variations are larger in GHG emissions compared to energy requirement. Results with economic allocation also show higher variability than mass- or energy-based allocation, due to higher variation of market prices. Percentage improvements over fossil fuels can be higher than 100% – meaning “negative” GHG emissions in the life-cycle – when the substitution method is used in the wheat chain. This method depends on the assumptions made with regard to avoided processes and thus may result in negative emissions when the credits of displaced processes are particularly high. Van der Voet et al. (2010) acknowledge that land use change is an important issue that should be included in life-cycle studies. Although one of the reviewed studies by van der Voet et al. (2010) takes into account nine different land use change scenarios for wheat cultivation (see Gnansounou et al. 2009), van der Voet et al. (2010) do not make any reference to the fact. If these

scenarios had been taken into account, the improvement in GHG emissions of wheat bioethanol over gasoline presented by van der Voet et al. (2010) would have changed from always positive (8% to 107%, cf. Table 3.2) to a much uncertain range including negative values (-120% to 107%), i.e. with potential higher GHG emissions of wheat bioethanol compared to gasoline.

Finally, Whitaker et al. (2010) performed a systematic review of, respectively, 18 and 15 wheat- and sugar beet-based bioethanol studies. Distinct sources of variation in results were identified, namely uncertainty in parameters (crop yields, fertilizer application rates, soil emissions), and methodological choices (co-product treatment approaches and format of data reporting). These authors also acknowledge the importance of calculating impacts under different land use change scenarios, but none of the wheat and sugar beet reviewed studies addressed the issue, which explains that all GHG intensity values listed in Table 3.2 are actually GHG savings over fossil fuels.

The present review identified ranges of energy requirement and GHG emissions for European wheat- and sugar beet-based bioethanol production systems. Several factors motivate the wide ranges shown, in particular which and how soil emissions are included, and the choice of calculation method to deal with co-products. All review studies recognize that direct land use change is an important issue in the GHG assessment of biofuel studies. However, only the review by Hoefnagels et al. (2010) quantifies the issue (for wheat). As a general conclusion, review studies in Table 3.2 stress the need to address the main sources of uncertainty and variability in biofuel life-cycle studies, in order to improve the robustness of calculations and increase the confidence in results.

Review study		Menichetti & Otto (2009) [a]	de Vries et al. (2010)	Hoefnagels et al. (2010)	van der Voet et al. (2010) [b]	Whitaker et al. (2010)
Relevant data, choices and assumptions						
No. of reviewed studies [c]:	wheat	6	9	3	6	18/11 (energy/GHG data)
	sugar beet	5	9	3	5	15/10 (energy/GHG data)
Temporal scale [d]:	wheat	2002 – 2007	2000 – 2007	2007 – 2009	2006 – 2009	n/d
	sugar beet	2002 – 2007	2002 – 2007	2007 – 2009	2005 – 2008	
Co-product credit approach		Subdivision + mass, energy, economic and mixed allocations + Substitution [e]	Substitution [f]	Mass, energy, and economic allocations + Substitution	Mass, energy, economic and carbon allocations + Substitution	Energy and economic allocations + Substitution
Carbon emissions from LUC (dLUC)		n/d	no	yes	no	no
N_2O emissions from land use		yes	yes	yes	n/d	yes
Indirect Land Use Change		no	no	no	no	no
Parameter uncertainty		yes [g]	n/d [h]	no	no	yes (N_2O emissions)
Selected results						
Energy Requirement E_{req} [MJ_p MJ_f^{-1}]		16% – 115% (wh) 24% – 85% (sb)	0.5; 0.67; 1.0 (wh) 0.53; 0.63; 0.77 (sb) (min; avg; max)	0.17 – 0.72 (wh) 0.04 – 0.63 (sb)	0.52 – 1.1 (wh) 0.63 – 0.67 (sb)	-0.71 – 0.50 (wh) [i] -0.52 – 0.56 (sb) [i]
GHG intensity [g CO_2eq MJ_f^{-1}]		18% – 90% (wh) 32% – 65% (sb)	34; 54; 74 (wh) 36; 46; 56 (sb) (min; avg; max)	21.8 – 80 (wh; no LUC) -244 – 247 (wh; LUC) [i] 11.0 – 66.2 (sb; no LUC) [j]	8% – 107% (wh) [a, k] 44% – 48% (sb) [a]	31 ± 6 to 61 ± 2 (wh) 45 ± 4 to 56 ± 5 (sb)

Table 3.2. Surveyed review studies of bioethanol production systems in Europe: relevant data and assumptions, methodological choices and key results.

Footnotes to Table 3.2: [a] Results are percentage improvements over fossil fuel equivalents; [b] Results available for one (sugar beet) and three (wheat) studies only; [c] Data presented in Table 3.2 relates to European biofuel production systems. Non-European chains, which were also addressed in some review studies, are excluded from this analysis; [d] Relates to the publication date of the reviewed studies; [e] These methods were applied in the reviewed studies, although it is not distinguishable which studies used which methods; [f] To improve comparability between reviewed studies, the authors used the substitution method to calculate co-product energy and GHG credits; [g] Concerning wheat and sugar beet, only one study considered error ranges for parameters; [h] The authors calculated average and standard deviation values based on the reviewed source publications. It was not possible to conclude however if any of the reviewed studies incorporated parameter uncertainty; [i] Very high credits were allocated to bioethanol production in specific scenarios, resulting in negative energy requirements and GHG emissions; [j] LUC results not available for sugar beet; n/d: not distinguishable; wh: wheat; sb: sugar beet; [k] These authors omit results of one of the reviewed studies in which several LUC scenarios were assessed. If not, percentage improvement in GHG intensity would have changed from 8% – 107% to -120% – 107%.

Reviewed study	**Menichetti & Otto (2009)**		**de Vries et al. (2010)**		**Hoefnagels et al. (2010)**		**van der Voet et al. (2010)**		**Whitaker et al. (2010)**	
Raw material	wheat	sugar beet	wheat	sugar beet	wheat	sugar beet	wheat	sugar beet	wheat	sugar beet
Börjesson (2009)							✔			
Gnansounou et al. (2009)							✔			
Smeets et al. (2009)					✔	✔				
Gabrielle & Gagnaire (2008)							✔			
Halleux et al. (2008)								✔		
JEC (2008)					✔	✔				
Renouf et al. (2008)								✔		
Bernard & Prieur (2007)							✔	✔		
Cocco (2007)										✔
DfT (2007)			✔	✔						
JEC (2007)	✔	✔	✔		✔	✔			✔	✔
Bernesson et al. (2006)									✔	
Fredriksson et al. (2006)							✔			
Malça and Freire (2006a)			✔	✔			✔	✔	✔	✔
Sims et al. (2006)									✔	✔
Lechón et al. (2005)	✔									
Powlson et al. (2005)									✔	✔
SenterNovem (2005b)	✔									
Tzilivakis et al. (2005)								✔		✔
Cariolle & Molard (2004)				✔						
Gnansounou & Dauriat (2004)		✔								
Mortimer et al. (2004)			✔	✔					✔	✔
Punter et al. (2004)			✔						✔	
Quirin et al. (2004)	✔									
Elsayed et al. (2003)	✔	✔	✔	✔					✔	✔
Woods & Bauen (2003)									✔	✔
ADEME (2002)	✔	✔	✔	✔					✔	✔
Armstrong et al. (2002)			✔	✔						
GM (2002)				✔					✔	✔
Rosenberger et al. (2001)									✔	
Richards (2000)			✔						✔	
Kaltschmitt et al. (1997)									✔	✔
Biewinga & van der Bijl (1996)									✔	✔
IEA (1994)									✔	✔
Ortiz-Canavate (1994)									✔	✔
Edwards et al. (1992)				✔						

Table 3.3. Wheat and sugar beet bioethanol studies addressed by the review works indicated in Table 3.2.

3.4. CONCLUDING REMARKS

The comprehensive reviews for biodiesel and bioethanol presented in this chapter show that several important issues affect the life-cycle energy and GHG balances of biofuel systems. Concerning biodiesel from rapeseed, a comprehensive review of 27 published life-cycle studies in Europe has been performed. A high variability of results, particularly for biodiesel GHG intensity, with emissions ranging from 15 to 170 g CO_2eq MJ^{-1} has been observed. The main causes for this high variability have been investigated, with emphasis on modeling choices. Key issues found are treatment of co-product and land use modeling, including high uncertainty associated with N_2O and carbon emissions from cultivated soil. Furthermore, a direct correlation between how soil emissions were modeled and increasing values for calculated GHG emission has been found for the surveyed studies.

This review also shows a time-dependent evolution of results: more recent assessments show higher GHG intensity and variability than former studies, due to evolving GHG modeling approaches used in biofuel life-cycle studies. Most former studies in this review show clear advantages for biodiesel over fossil diesel in terms of life-cycle GHG intensity. Moreover, these studies report a correlation between biodiesel nonrenewable energy inputs and GHG emissions. Other studies in the literature point out the same conclusion. However, this chapter demonstrates that taking into account soil emissions in biofuel life-cycle assessments, namely N_2O emissions due to land use and carbon emissions due to land use change, negates that correlation. Soil emissions are not exclusively linked to energy use; hence, energy cannot be used as a proxy for emissions. The review also shows that soil emissions take the lead over energy use in terms of the critical factor for the overall GHG intensity of biodiesel. In particular, taking account of parameter uncertainty for soil emissions strongly affects the GHG emission results of biodiesel.

Concerning bioethanol, the five review works – which altogether address 36 different life-cycle studies for wheat and sugar beet bioethanol – do not include the contribution of dLUC in the results, with the exception of the review by Hoefnagels et al. (2010).

Therefore, energy and GHG intensity results follow similar trends and indicate net savings over gasoline. Only Hoefnagels et al. (2010) demonstrate that the inclusion of dLUC, particularly when land use change induce high carbon release from soils, may significantly deviate the GHG balance from the energy balance for the same biofuel chain. The most recent study of the 29 life-cycle studies covered in the review also takes into account soil carbon exchange due to direct land use change, demonstrating that this issue is of utmost importance in the GHG intensity (and its uncertainty) of the wheat bioethanol chain.

This chapter highlights the need for transparency in assumptions and inputs to life-cycle models and demonstrates that neglecting key issues – and related uncertainty – in the life-cycle GHG accounting of biofuels may compromise the reliability of results. It is important to incorporate uncertainty analysis in the life-cycle modeling of biofuels, in order to reduce the uncertainty level in the results and to better support decisions on whether or not to promote specific biofuel pathways. Emissions from cultivated soils have a substantial effect on biofuel GHG intensity and require further research efforts to improve. One of the most influential issues raised in this review and often overlooked in the life-cycle assessments available in the literature – soil carbon emissions from land use change – is further discussed in chapter 5.

4. Modeling Biofuel Systems

"We can get fuel from fruit, from that shrub by the roadside, or from apples, weeds, saw-dust – almost anything! There is fuel in every bit of vegetable matter that can be fermented. There is enough alcohol in one year's yield of a hectare of potatoes to drive the machinery necessary to cultivate the field for a hundred years. And it remains for someone to find out how this fuel can be produced commercially – better fuel at a cheaper price than we know now."

Henry Ford, 1925

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4. MODELING BIOFUEL SYSTEMS

4.1. PURPOSE AND SCOPE

The motivation and scientific background to the core of this dissertation have been already presented in chapters 1 and 2. At first, to demonstrate the application of the methodology, namely the implications of incorporating uncertainty in the life-cycle modeling of biofuels, any biofuel production system would be appropriate. This chapter provides a detailed description of first-generation biofuel systems brought from the European context: vegetable oil and biodiesel from rapeseed; bioethanol from wheat and sugar beet, and its derivative bioETBE. The rationale for choosing these chains has been threefold:

- firstly, these biofuel chains are sufficiently diverse to enable the demonstration of several important points when addressing uncertainty issues;
- secondly, these chains represent the majority of biofuel systems (production and consumption) at the European level, which is the geographical scope of this investigation (EurObserv'ER 2008, 2009, 2010);

- thirdly, it is expected that in coming years (at least in the next decade) first-generation biofuels still represent a major share at the European biofuels market (Bowyer 2010).

This chapter is organized as follows. Firstly, data on production and use of biofuels in Europe is presented, including historic and legal framework aspects. Secondly, major chain steps are described, with emphasis on discussing how scenario and parameter uncertainty issues have been taken into account. In particular, multifunctionality options for each chain are discussed. Collated life-cycle inventory data and computed probability density functions representative of European biofuel production systems are presented. Thirdly, a sensitivity analysis to evaluate the response of modeled biofuel chains to variations in individual parameters is presented. Finally, practical aspects concerning the application of Monte-Carlo simulation are discussed.

4.2. FIRST-GENERATION BIOFUELS IN EUROPE

PRODUCTION AND 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. Plant oils can also be blended with petroleum diesel or converted into a petroleum diesel substitute (biodiesel), through a transesterification reaction with an alcohol, usually methanol.

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). Nowadays, 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 (Chiaramonti and Tondi 2003). Vegetable oils are also used after conversion to biodiesel. The European Union holds the leading position at worldwide level in terms of biodiesel production (EBB 2009). Germany and France are the main biodiesel producers, with a share of nearly 50% of total production in 2009 (EurObserv'ER 2010). The most used raw material is rapeseed, accounting for nearly 84% of the total European biodiesel feedstock (Fischer et al. 2009). In 2009, biodiesel reached a market share of approximately 3.2% in terms of total fuel consumption in the European transportation sector, or 4.4% if compared with fossil diesel consumption (assuming a 2.5 ratio of diesel to gasoline consumption in Europe, according to Eurostat 2009). Table 4.1. gathers information regarding biodiesel consumption and market shares in recent years in the EU-27 (Malça and Freire 2011a).

Table 4.1. Biodiesel consumption for transport in the EU-27, including market shares and major biodiesel consumers (Malça and Freire 2011a).

Biodiesel consumption, ktoe [(1)]	**2005**	**2006**	**2007**	**2008**	**2009**
Germany	1548	2532	2906	2382	2224
France	344	589	1214	1859	2056
United Kingdom	25	132	271	698	823
Italy	172	149	136	658	1049
Spain	23	54	259	520	894
Total biodiesel (EU-27), ktoe	**2245**	**4074**	**5899**	**8018**	**9616**
Yearly growth, ktoe (%)	-	1829 (81.4%)	1825 (44.8%)	2119 (35.9%)	1598 (19.9%)
Total biofuel (EU-27), ktoe	**2991**	**5376**	**7834**	**10189**	**12093**
Biodiesel share, %	75.1	75.8	75.3	78.7	79.5
Biofuels' incorporation rate [(2)], %	1.0	1.8	2.6	3.3	4.0

[(1)] ktoe: thousand tonnes of oil equivalent;

[(2)] Biofuel incorporation rate in energy content of total fuel consumption in the transportation sector.

Biological feedstocks that contain appreciable amounts of sugar (e.g. sugar beet and sugar cane) and materials that can be converted into sugar, such as starch (e.g. wheat, corn and barley), may be fermented to produce bioethanol, commonly used in spark-ignition engines (and to a lesser extent in compression-ignition engines). Bioethanol can also be used as feedstock to produce bio-Ethyl Tertiary Butyl Ether (bioETBE), through the chemical reaction of bioethanol with isobutylene (by-product of the petroleum refining process).

Bioethanol has been known as a motor fuel for many decades. Its use for transport started in the beginning of the XX century, when Henry Ford designed the Ford model T in the expectation that bioethanol produced by American farmers would be used as its primary fuel. Nevertheless, petroleum fuel became the major transportation fuel since the early XX century because of the better compatibility with the materials used at the time for engine manufacturing and also the growing supply of cheaper fuel from oil field discoveries. The use of ethanol as a transportation fuel was mostly abandoned after World War II and was only resumed in the early 1970´s, when the first world oil crisis motivated the search for fuel alternatives to petroleum. In Europe, ethanol fuel production started in 1992, but the sector has only soared after 2004. From 2004 to 2009, ethanol fuel production almost septupled, reaching 3600 million liters in 2009 (ePURE 2011). Major European producers of bioethanol fuel are France, Germany and Spain, which represented almost 70% of the EU-27 production in 2009 (ePURE 2011). According to the European Renewable Ethanol Association (ePURE 2011), nearly two thirds of the current bioethanol production in Europe is based on cereals, and wheat is the most commonly used feedstock. The remaining one third comes from sugar beet and molasses[1]. Table 4.2 shows the EU consumption of bioethanol in the past 5 years. In several countries, e.g. France and Spain, bioethanol is converted to bioETBE and may be used in proportions of up to 22% by volume.

[1] Molasses is also a raw material for the production of bioethanol. It is a byproduct of sugar production from sugar beet (beet molasses) and sugarcane (cane molasses) (Mojovic et al. 2009).

Table 4.2. Bioethanol fuel consumption for transport in the EU-27, including market shares and major bioethanol consumers (calculated from EurObserv'ER 2007-10 data).

	2005	2006	2007	2008	2009
Germany	144.6	304.7	296.5	403.7	581.7
France	74.9	147.8	272.1	414.6	455.9
Sweden	144.5	162.9	182.2	214.9	199.4
UK	43.1	48.5	77.9	103.3	159.0
Spain	113.0	114.5	130.0	93.2	152.2
Total bioethanol (EU-27)	557.3	871.7	1200.5	1773.8	2339.2
Yearly growth (ktoe (%))	-	314.4 (56.4%)	328.8 (37.7%)	573.3 (47.8%)	565.4 (31.9%)
Total biofuel (EU-27)	2992.0	5601.7	7834.2	10189.1	12092.6
Bioethanol share (%)	18.6	15.6	15.3	17.4	19.3

EUROPEAN LEGAL FRAMEWORK. In May 2003, the EU adopted a directive on the promotion of the use of biofuels or other renewable fuels for transport (EPC 2003). According to this directive, Member States should ensure that a minimum proportion of biofuels and other renewable fuels is placed on their markets. Specific targets have been set for years 2005 and 2010, respectively 2% and 5.75%, calculated on the basis of the energy content of all gasoline and diesel marketed for transport purposes. Later, in January 2007, the European Commission proposed "An energy policy for Europe", with the goal to combat climate change and boost the EU's energy security and competitiveness (EC 2007). Not only 98% of the EU transport sector relies on oil, but also about 80% of EU oil demand is met by imports. Moreover, transport GHG emissions increased by almost 30% since 1990, a trend that is opposed to all other sectors covered by the Kyoto Protocol. Based on the European Commission's proposal, in March 2007 the Council endorsed the target of raising the share of biofuels in the transport sector to 10% by 2020. Nevertheless, growing concerns in recent years that the production of biofuels might not respect minimum environmental and social requirements lead to the publication of Directive 2009/28/EC (EPC 2009) on the promotion of the use of energy from renewable sources (Renewable Energy Directive, or RED directive). Influenced by the potential negative impacts of biofuels, the EU has broadened the 10% biofuel target: apart from biofuels other renewable energy carriers, such as electricity or hydrogen, may contribute as well to the target. Moreover, compliance with the targets laid down

in the directive is only considered for biofuel pathways for which the fulfillment of specific sustainability criteria is demonstrated.

Nevertheless, in a recently published study that analyzed the NREAPs[2] of EU Member States, it is anticipated that first-generation biofuels will remain as the main source for delivering the RED 2020 target on biofuels (Bowyer 2010). It is estimated that approximately 9.5% of the energy in the transport sector will be sourced from biofuels in 2020 (within a total target of 10%), of which more than 90% will be fulfilled through first-generation biofuels.

4.3. RAPESEED OIL AND RAPESEED METHYL ESTER (BIODIESEL)

The life-cycle stages of the Rapeseed Oil (RO) and Rapeseed Methyl Ester (RME, biodiesel) chains include rapeseed cultivation, harvesting, transport and drying of the seeds, crushing and extraction of the oil, oil degumming and refining. The final step is the chemical reaction (transesterification) to convert RO into RME. These steps are illustrated in the flowchart of Fig. 4.1. A detailed description of the RO and RME production systems can be found, for example, in Mortimer and Elsayed (2006), Stephenson et al. (2008) and Malça and Freire (2009a, 2010a).

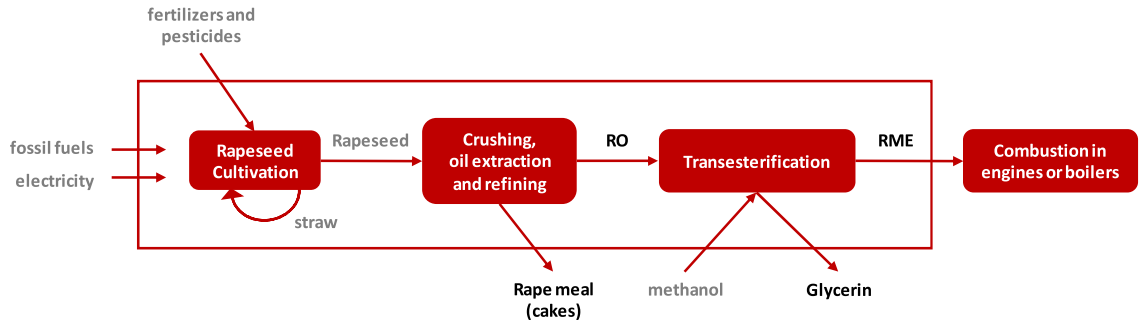


Fig. 4.1. Flow chart illustrating the RO and RME production chains. Transportation activities and optional blending with petroleum diesel are not shown for the sake of simplicity (RO: Rapeseed Oil; RME: Rapeseed Methyl Ester).

[2] In the National Renewable Energy Action Plans (NREAPs), European governments specify how they plan to deliver their targets under the Renewable Energy Directive (Directive 2009/28/EC).

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 plowed back into the field. 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 (SenterNovem 2005a; JEC 2007; UFOP 2008; Börjesson and Tufvesson 2011).

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.

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.

In the transesterification reaction (Fig. 4.2), the triglyceride molecules of the oil are reacted with methanol in the presence of an alkaline catalyst (usually sodium methylate, potassium hydroxide or sodium hydroxide, to improve the reaction rate and yield), producing a mixture of rapeseed methyl ester and glycerin (Ma and Hanna 1999). The transesterification process involves three reversible reactions, whereby the triglyceride is converted successively to diglyceride, monoglyceride and glycerol, consuming one mole of alcohol in each step and liberating one mole of ester (Stephenson et al. 2008). After settling, glycerin is left on the bottom and RME is left on top. Finally, RME is recovered, washed and filtered (Demirbas 2001). The purpose of transesterification is to lower the viscosity of the oil, improving combustion in diesel engines (Basha et al. 2009). Thorough reviews of the use of biodiesel as alternative fuel for diesel engines can be found e.g. in Shahid and Jamal (2008) and Murugesan et al. (2009).

Some authors assume that tailpipe CO_2 emissions from RME combustion are neutral, being balanced by the CO_2 sequestered during crop growth, which does not occur for fossil diesel. If the methanol feedstock used in the transesterification reaction is produced from fossil fuels, then the carbon atoms from methanol contribute to emissions that were not previously offset and must be taken into account (Bernesson et al. 2004; Wicke et al. 2008; Thamsiriroj and Murphy 2009).

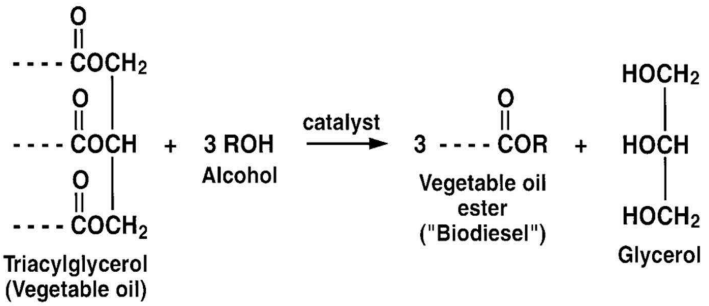


Fig. 4.2. The transesterification reaction (Knothe 2001). The most used alcohol in this step – methanol – comes mainly from fossil resources, which detracts from the sustainability of biodiesel production. Dashed line in molecular structures represents fatty acid chains.

The multifunctionality of biofuel systems is considered a critical issue in biofuel life-cycle studies, as discussed before. For the RME production system, in particular, two valuable co-products are obtained: rape meal and glycerin, as illustrated in Fig. 4.1. Different approaches may be addressed for dealing with this co-production (ISO 14044:2006; Guinée et al. 2009): (i) the substitution method, in which the system is expanded with avoided processes – soy meal production and several glycerin potential uses – to remove additional functions related to the functional flows; (ii) allocation, or partitioning, of the RME multifunctional process, i.e. splitting up the process into single-functional processes (RME production + rape meal production + glycerin production) on the basis of underlying relationships (physical: mass, energy; and economic); and (iii) the no allocation or surplus method, in which the additional functional flows – rape meal and glycerin – that are not strictly needed for the RME system under study are ignored, i.e. all burdens (energy and material inputs, and related emissions) are allocated to RME.

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).

Currently utilized options for glycerin include its use as a supplement for animal feed or as a boiler fuel (Johnson and Taconi 2007). Several works have demonstrated the ability of glycerin for animal feed: crude glycerin provides a highly available energy source for growing pigs and laying hens, with a metabolizable energy content 14% higher than that of corn grain (Lammers et al. 2008a, 2008b, 2008c). Pigs, for example, can be fed up to 10% crude glycerin, according to Lammers et al. (2008a). In Germany, both crude and refined glycerin are allowed as feed ingredients; it is common to use between 2% and 5% of crude glycerin in the animal feed mix for poultry and pigs (Hoogendoorn et al. 2007). Glycerin from biodiesel production may also be used for energy in heat

production boilers, displacing conventional fuels; this approach is used e.g. by Soimakallio et al. (2009) and GM (2002). Johnson and Taconi (2007) provide a review of promising options for both the catalytic and biological conversion of crude glycerin into a wide variety of value-added products, many of which are bio-based alternatives to petroleum-derived chemicals. Although these alternatives present a substantial market improvement for crude glycerin, many of the technologies involved still need additional research and development to make them economically and operationally feasible for incorporation into existing biorefineries (Johnson and Taconi 2007). One of many chemicals that glycerin might displace is propylene glycol in antifreeze (JEC 2007; Johnson and Taconi 2007; Lechón et al. 2009).

Crude glycerin may also be refined to pharmaceutical grade glycerin, so that it can replace synthetic glycerin. However, the tight supply and demand market for synthetic glycerin cannot accommodate the excess amounts generated from biodiesel production: replacing just 5% of petroleum diesel with biodiesel would result in a glycerin production of more than 30 times the EU production of synthetic glycerin (JEC 2007; Johnson and Taconi 2007). As already discussed in chapter 2, market restrictions for a potential replacement scenario are one of the limitations of the substitution method and should be linked to a specified amount of biofuels produced.

Concerning the substitution method, this dissertation assumes that rapeseed meal displaces soy meal imports. Amongst the multiple alternatives for glycerin, the substitution options considered in this dissertation include replacing grain as animal feed, for process heat, and displacing propylene glycol or synthetic glycerin. Even acknowledging that substitution credits for co-products should only be applied if the co-product is actually accepted by the market as a replacement, the substitution option for synthetically produced glycerin is considered in this dissertation, as it represents an upper limit for energy and emission credits given to the RME chain (Wicke et al. 2008). Actually, synthetic glycerin production has a fossil fuel requirement of approximately 18 times its heating value, which represents a very favorable credit (JEC 2007). Table 4.3 gathers data concerning synthetic glycerin production and the additional crude glycerin refining step.

To build life-cycle inventory tables for European RO and RME production systems, data collection has been conducted in several sources, namely commercial databases, scientific articles and technical reports. Table 4.3 gathers the information concerning parameter values used in the life-cycle of rapeseed oil and RME. Some aspects of the life-cycle inventory process follow:

- in some literature sources data is highly disaggregated whereas in others only total figures are provided, which makes difficult to compare processes and justify differences in data;
- differences concerning similar processes may be justified by the geographic and time scales used, and/or by the technologies employed;
- data from different literature sources have been converted to common units using (preferably) information provided in the same source;
- although data collection has been focused on characterizing typical biofuel production systems in Europe, it was not possible however to estimate the representativeness of each data point due to lack of information.

Table 4.4 lists the probability density functions that have been selected to fit input parameters in the RO/RME production system. The practical rules to select the most appropriate distributions have already been described in chapter 2. Data concerning feedstock production, as well as generic data for biofuel life-cycle modeling are provided in appendix (Table A.1).

Table 4.3. Parameter values considered in the RO and RME life-cycle inventory.

Parameter [(1)]				**Source**
Seed rate	*Min*	*Med*	*Max*	
[kg ha^{-1} yr^{-1}]		7		*Richards (2000)*
		5		*Elsayed et al. (2003)*
		5		*Elsgaard (2010)*
		5		*Brandão et al. (2011)*
N fertilizer application rate	*Min*	*Med*	*Max*	
[kg ha^{-1} yr^{-1}]		200		*Epelly (1993)*
	80		185	*FAO (2002)*
		140		*Bernesson et al. (2004)*
		187		*McManus et al. (2004)*
	195		225	*SenterNovem (2005a)*
	60	135	200	*Lecomte (2006)*
		184		*Mortimer & Elsayed (2006)*
		144		*Halleux et al. (2008)*
	160		211	*Stephenson et al. (2008)*
		188		*ADEME (2010)*
		166		*de Vries et al. (2010)*
	166		186	*Elsgaard (2010)*
				Börjesson&Tufvesson
		145		*(2011)*
		196		*Brandão et al. (2011)*
P_2O_5 fertilizer application rate[kg ha^{-1} yr^{-1}]		60		*Epelly (1993)*
	53		78	*GM (2002)*
		34		*Bernesson et al. (2004)*
		70		*McManus et al. (2004)*
	45	60	75	*SenterNovem (2005a)*
		55		*Mortimer & Elsayed (2006)*
		74		*Halleux et al. (2008)*
		61		*Stephenson et al. (2008)*
		44		*ADEME (2010)*
		53		*Elsgaard (2010)*
				Börjesson&Tufvesson
		44		*(2011)*
		50		*Brandão et al. (2011)*
K_2O fertilizer application rate		120		*Epelly (1993)*
[kg ha^{-1} yr^{-1}]	30		180	*GM (2002)*
		30		*Bernesson et al. (2004)*
		130		*McManus et al. (2004)*
	30	40	50	*SenterNovem (2005a)*
		64		*Mortimer & Elsayed (2006)*
		74		*Halleux et al. (2008)*
		72		*Stephenson et al. (2008)*
		33		*ADEME (2010)*
		107		*Elsgaard (2010)*
				Börjesson&Tufvesson
		12		*(2011)*
		48		*Brandão et al. (2011)*
Pesticides application rate		1.5		*Epelly (1993)*
[kg ha^{-1} yr^{-1}]		4.75		*Ceuterick & Spirinckx (1997)*
		1.22		*GM (2002)*
		2.11		*McManus et al. (2004)*
		2.15		*Bernesson et al. (2004)*
		2.8		*Mortimer & Elsayed (2006)*
		2.3		*Halleux et al. (2008)*
		2.4		*de Vries et al. (2010)*

		1.68		*Elsgaard (2010)*
		2.8		*Brandão et al. (2011)*
Fuel consumption of agricultural machinery (diesel) [liter ha^{-1}]		76		*Epelly (1993)*
		63.6		*GM (2002)*
		40.2		*McManus et al. (2004)*
		65.9		*Bernesson et al. (2004)*
		150		*SenterNovem (2005a)*
		81-93		*Dalgaard & Dalgaard (2006)*
		66.3		*Mortimer & Elsayed (2006)*
		131.7		*Halleux et al. (2008)*
		59.5 ± 10%		*Elsgaard (2010)*
N_2O emissions from soil [kg N_2O kg^{-1} N fert]		*default*	*range*	
			0.007 – 0.041	*Kaiser et al. (1998)*
				Ecobilan (1999)
		0.0196	0.0 – 0.0471	*Patyk & Reinhardt (2000)*
		0.01	0.004 – 0.017	*Richards (2000)*
		1.6 – 7.8 kg N_2O ha^{-1}		*Freibauer&Kaltschmitt (2001)*
		0.005		*ADEME (2002)*
		0.030	0.0046 – 0.084	*GM (2002)*
		0.0036		*Mortimer et al. (2003)*
		0.0196		*Bernesson et al. (2004)*
		0.0053		*McManus et al. (2004)*
			0.016 – 0.035	*SenterNovem (2005a)*
			0.7–0.8 kg N_2O ha^{-1}	*Tzilivakis et al. (2005)*
		0.004	0.0025 – 0.0225	*Lechón et al. (2006)*
		0.0237		*Mortimer & Elsayed (2006)*
		0.0208	0.00489 – 0.102	*IPCC (2006)* [(2)]
		3.12 kg N_2O ha^{-1}	1.89 – 4.35 kg N_2O ha^{-1}	*JEC (2007)* [(3)]
			0.016 – 0.035	*Zah et al. (2007)*
		0.047 – 0.079		*Crutzen et al. (2008)*
			0.015 – 0.05	*Reijnders&Huijbregts (2008)*
		0.022		*Stephenson et al. (2008)*
		0.004		*Lechón et al. (2009)*
		0.0255	0.004 – 0.112	*Soimakallio et al. (2009)*
		4.55 kg N_2O ha^{-1}		*FA (2010)*
		0.01 – 0.0175		*GA (2010)*
		0.021		*Börjesson&Tufvesson (2011)*
			0.0063 – 0.047	*Brandão et al. (2011)*
Agricultural yield [t ha^{-1}]	*Min*	*Med*	*Max*	
	2.5		3.5	*Dupic (1994)*
	3	4	5	*SenterNovem (2005a)*
		3.1		*Prolea (2007)*
		3.3		*de Vries et al. (2010)*
	2.6		4.3	*FAOSTAT (2010)* [(4)]
Oil extraction rate [%]		39		*Epelly (1993)*
		> 40		*Dupic (1994)*
		39		*Schöpe & Britschkat (2002)*
		40		*McManus et al. (2004)*
		40 - 45		*SenterNovem (2005a)*
		39 – 41		*FA (2010)*
		37		*de Vries et al. (2010)*
Transportation activities				
Distances traveled, single trip [km]				
From farms to processing facilities		200 ± 50%		*Poitrat et al. (1998)*
		260		*Stephenson et al. (2008)*

From plants to local distribution depots	200 ± 50%	*Poitrat et al. (1998)*
	240	*Stephenson et al. (2008)*
RO production (feedstock)		
Hexane [kg t^{-1} RO]	3.4	*Epelly (1993)*
Hexane [kg t^{-1} RO]	2.5	*Elsayed et al. (2003)*
Hexane [kg t^{-1} RO]	2.4	*Mortimer & Elsayed (2006)*
Hexane [kg t^{-1} RME]	2.7	*Halleux et al. (2008)*
Hexane [kg t^{-1} RME]	2.4	*Stephenson et al. (2008)*
Phosphoric Acid [kg t^{-1} RO]	0.75	*Epelly (1993)*
Phosphoric Acid [kg t^{-1} RO]	1.0	*Elsayed et al. (2003)*
Phosphoric Acid [kg t^{-1} RME]	1.0	*Stephenson et al. (2008)*
Sodium Hydroxide [kg t^{-1} RO]	1.4	*Epelly (1993)*
Sodium Hydroxide [kg t^{-1} RME]	6.0	*Elsayed et al. (2003)*
Sodium Hydroxide [kg t^{-1} RO]	0.6	*Mortimer & Elsayed (2006)*
Sodium Hydroxide [kg t^{-1} RME]	3.0	*Stephenson et al. (2008)*
RME production (feedstock)		
Methanol [kg t^{-1} RME]	110	*Epelly (1993)*
	109	*Elsayed et al. (2003)*
	113	*Mortimer & Elsayed (2006)*
	109	*Halleux et al. (2008)*
	110	*Stephenson et al. (2008)*
	95	*ADEME (2010)*
Sulphuric acid [kg t^{-1} RME]	5.6	*Epelly (1993)*
	20	*Mortimer & Elsayed (2006)*
	10.4	*Stephenson et al. (2008)*
Alkaline catalyst [kg t^{-1} RME]	4.6	*Epelly (1993)*
	23.4	*Mortimer & Elsayed (2006)*
	10.3	*Stephenson et al. (2008)*
RO production (energy use)		
Electricity (grain drying) [MJ t^{-1} rapeseed]	55.3	*Epelly (1993)*
Electricity (grain drying) [MJ t^{-1} rapeseed]	41.8	*Elsayed et al. (2003)*
Electricity (grain drying) [MJ t^{-1} rapeseed]	18.5	*Mortimer & Elsayed (2006)*
Electricity (grain drying) [MJ t^{-1} RME]	119	*Halleux et al. (2008)*
Electricity (grain drying) [MJ t^{-1} rapeseed]	45 – 75	*Stephenson et al. (2008)*
Fuel oil (grain drying) [MJ t^{-1} rapeseed]	305	*Elsayed et al. (2003)*
Fuel oil (grain drying) [MJ t^{-1} rapeseed]	132	*Mortimer & Elsayed (2006)*
Heat (grain drying) [MJ t^{-1} RME]	812	*Halleux et al. (2008)*
Fuel oil (grain drying) [MJ t^{-1} rapeseed]	143 – 218	*Stephenson et al. (2008)*
Electricity (oil extraction) [MJ t^{-1} RO]	369	*Epelly (1993)*
Electricity (oil extraction) [MJ t^{-1} RO]	302	*Elsayed et al. (2003)*
Electricity (oil extraction + refin) [MJ t^{-1} RO]	455	*Mortimer & Elsayed (2006)*
Electricity (oil extraction) [MJ t^{-1} RME]	382	*Halleux et al. (2008)*
Electricity (oil extraction) [MJ t^{-1} RO]	1214	*Stephenson et al. (2008)* [(5)]
Natural Gas (oil extraction) [MJ t^{-1} RO]	1846	*Epelly (1993)*
Natural Gas (oil extraction) [MJ t^{-1} RO]	1790	*Elsayed et al. (2003)*
Natural Gas (oil extraction + refin) [MJ t^{-1} RO]	2354	*Mortimer & Elsayed (2006)*
Heat (oil extraction) [MJ t^{-1} RME]	2317	*Halleux et al. (2008)*
Natural Gas (oil extraction) [MJ t^{-1} RO]	2616	*Stephenson et al. (2008)* [(5)]
Electricity (oil refining) [MJ t^{-1} RO]	18	*Epelly (1993)*
Electricity (oil refining) [MJ t^{-1} RO]	11.2	*Elsayed et al. (2003)*
Electricity (oil refining) [MJ t^{-1} RME]	40	*Halleux et al. (2008)*
Electricity (oil refining) [MJ t^{-1} RO]	339	*Stephenson et al. (2008)*
Natural Gas (oil refining) [MJ t^{-1} RO]	324	*Epelly (1993)*
Natural Gas (oil refining) [MJ t^{-1} RO]	178	*Elsayed et al. (2003)*
Heat (oil refining) [MJ t^{-1} RME]	162	*Halleux et al. (2008)*
Natural Gas (oil refining) [MJ t^{-1} RO]	559	*Stephenson et al. (2008)* [(5)]
RME production (energy use)		
Electricity (transesterification) [MJ t^{-1} RME]	460	*Epelly (1993)*
Electricity (transesterification) [MJ t^{-1} RME]	83	*Elsayed et al. (2003)*
Electricity (transesterification) [MJ t^{-1} RME]	335	*Mortimer & Elsayed (2006)*
Electricity (transesterification) [MJ t^{-1} RME]	133	*Halleux et al. (2008)*

Electricity (transesterification) [MJ t^{-1} RME]		543		*Stephenson et al. (2008)*
Natural Gas (transesterification) [MJ t^{-1} RME]		1840		*Epelly (1993)*
Natural Gas (transesterification) [MJ t^{-1} RME]		1402		*Elsayed et al. (2003)*
Natural Gas (transesterification) [MJ t^{-1} RME]		2851		*Mortimer & Elsayed (2006)*
Heat (transesterification) [MJ t^{-1} RME]		947		*Halleux et al. (2008)*
Natural Gas (transesterification) [MJ t^{-1} RME]		1671		*Stephenson et al. (2008)*
RO/RME production (conversion ratios)				
Rape meal [t t^{-1} RME]		1.60		*Epelly (1993)*
		1.58		*Halleux et al. (2008)*
RME [t t^{-1} RO]		0.978		*Epelly (1993)*
Glycerin [kg t^{-1} RME]		102		*Epelly (1993)*
		100		*Halleux et al. (2008)*
Co-product substitution ratio		*kg soy meal kg^{-1} rape meal*		
Rape meal vs. Soy meal		0.70		*Scharmer (2001)*
		0.80		*JEC (2007)*
		0.77		*Soimakallio et al. (2009)*
		0.85		*Börjesson&Tufvesson (2011)*
Substitution credits				
Rape meal vs. Soy meal		*MJ kg^{-1} soy meal*	*$kgCO_2eq\ kg^{-1}$ soymeal*	
		5.1	1.38	*UBA (1999)*
		3.7	0.68	*Scharmer (2001)*
		5.6	0.50	*JEC (2007)*
		3.5	0.065	*JEC (2007)* [6]
			0.55 ± 50%	*Wicke et al. (2008)*
		9.7 ± 30%	0.23 ± 30%	*Soimakallio et al. (2009)*
		9.3	0.98	*Börjesson&Tufvesson (2011)*
Glycerin for animal feed		*MJ kg^{-1} RME*	*$kgCO_2eq\ kg^{-1}$RME*	
		0.23	0.032	*JEC (2007)*
Glycerin for process heat		*MJ kg^{-1} RME*	*$kgCO_2eq\ kg^{-1}$RME*	
		1.39	0.07	*GM (2002)*
Glycerin for propylene glycol		*MJ kg^{-1} prop. glyc.*	*$kgCO_2eq\ kg^{-1}$ prop glyc*	
		13.4	2.17	*JEC (2007)*
Synthetic glycerin production		*MJ kg^{-1} glycerin*	*kg $CO_2eq\ kg^{-1}$ glycerin*	
		177	10.4	*UBA (1999)*
		127	5.6	*Patyk & Reinhardt (2000)*
		209	9.0	*Scharmer (2001)*
		161	11.0	*GM (2002)*
		180	10.5	*JEC (2004)*
		134	9.6	*GEMIS (2005)*
Crude glycerin refining		*MJ kg^{-1} glycerin*	*kg $CO_2eq\ kg^{-1}$ glycerin*	
		10.4	0.82	*Rollefson et al. (2004)*
		15.7	0.94	*JEC (2007)*
		7.4	0.58	*Lurgi (2008)*
Market prices [€ t^{-1}]	*Min*	*Med*	*Max*	
Rapeseed Oil	409	511	614	*Schöpe&Britschkat (2002)* [7]
	422	524	626	*Schöpe&Britschkat (2002)* [8]
		485		*Mortimer & Elsayed (2006)*
RME		579	724	*Schöpe & Britschkat (2002)*
		679		*Bernesson et al. (2004)*
	667	1011	1069	*Börjesson&Tufvesson (2011)* [9]
		966		*DGEG (2011)*
Rape Meal	86	105	124	*Schöpe&Britschkat (2002)*
	149	191	199	*Bernesson et al. (2004)*
		126		*Mortimer & Elsayed (2006)*
	180	240	260	*Börjesson&Tufvesson*

				(2011) [9]
Glycerin		476		*Bernesson et al. (2004)*
	180	360	540	*Börjesson&Tufvesson (2011)* [9]
	70	100	200	*APPB (2011)*

[1] – Energy and GHG emissions of fertilizer and pesticides production, as well as generic data for biofuel life-cycle modeling are provided in appendix (Table A.1);
[2] – These average and uncertainty ranges for N_2O emissions are own calculations based on the Tier 1 methodology provided by the IPCC (2006). This is a straightforward method that combines readily available national statistics with emission factors. Direct N_2O emissions due to N inputs to managed soils and indirect N_2O emissions due to volatilization of N as NH_3 and NO_x and leaching and runoff of N from soils have been calculated using the default values and uncertainty ranges provided in IPCC (2006). There is no information regarding the statistical significance of the uncertainty ranges in IPCC (2006), thus calculated ranges should be interpreted carefully;
[3] – JEC (2007) estimates N_2O emissions from soils using the database-calculation model developed by the Joint Research Centre of the EU (JRC) in combination with the soil chemistry model of the Univ. of New Hampshire (DNDC model). Although an effort has been made to include crop and soil characteristics based on a land-cover survey for the EU-15, the lack of data concerning indirect N_2O emissions required the use of IPCC (2006) emission factors for these emissions. It must be emphasized that there is still significant uncertainty in the estimation of N_2O emissions in JEC (2007) and that no information is provided concerning the statistical significance of the computed ranges. Moreover, measurements in individual fields may by far surpass the emissions predicted by these models;
[4] – Based on top five European producers (2005-2009 data);
[5] – Energy consumption as primary energy;
[6] – Taking into account energy and GHG credits of soy oil (co-produced with soy meal) displacing rapeseed oil;
[7] – Prices when oil extraction and transesterification take place in nearby infrastructures;
[8] – Prices when transesterification plants are not in the vicinity of oil mills;
[9] – Estimated average prices for 2008.

Table 4.4. RO and RME production systems: Probability distributions for input data and characteristic values computed for each distribution.

Parameter	distribution	min	mean	max	std dev	scale	shape
Fertilizer application rates [kg ha^{-1} yr^{-1}]							
N fertilizer	Normal		150		15		
P_2O_5 fertilizer	Normal		55		5.5		
K_2O fertilizer	Lognormal		75		20		
Seed application rate [kg ha^{-1} yr^{-1}]	Uniform	5		7			
Pesticides application rate [kg ha^{-1} yr^{-1}]	Normal		2.4		0.36		
Fuel consumption agricultural machinery [l ha^{-1}]	Lognormal		76		22.8		
Soil carbon stock changes [ton C ha^{-1} yr^{-1}] [1]							
N_2O emissions from soil [kg N_2O kg^{-1} N fert]	Lognormal		0.0208		0.021		
Agricultural yield [ton ha^{-1}]	Normal		3.5		0.18		
Oil extraction rate [%]	Weibull	39				3	2
Rapeseed transport [km]	Normal		200		30		
Rapeseed oil transport [km]	Normal		200		30		
RME transport (road/rail) [km]	Normal		250		37.5		
Energy use in generic industrial processes [2]	Normal		[2]		Mean/20		
RO production							
Electricity (grain drying) [MJ t^{-1} rapeseed]	Normal		50		2.5		
Natural Gas (grain drying) [MJ t^{-1} rapeseed]	Normal		200		10		
Electricity (oil extraction) [MJ t^{-1} RO]	Normal		270		13.5		
Natural Gas (oil extraction) [MJ t^{-1} RO]	Normal		1400		70		
Electricity (oil refining) [MJ t^{-1} RO]	Normal		15		0.75		
Natural Gas (oil refining) [MJ t^{-1} RO]	Normal		260		13		
RME production							
Electricity (transesterification) [MJ t^{-1} RME]	Normal		360		18		

Natural Gas (transesterification) [MJ t^{-1} RME]	Normal		1500	75		
Substitution credits						
Ratio soy meal / rape meal [kg kg^{-1}]	Normal		0.78	0.039		
Energy use in soy meal production [MJ kg^{-1}]	Normal		3.53	0.29		
Emissions in soy meal production [kg CO_2eq kg^{-1}]	Normal		0.065	0.097		
Energy use glycerin animal feed [MJ kg^{-1} RME]	Normal		0.23	0.011		
Emissions glycerin animal feed [kg CO_2eq kg^{-1} RME]	Normal		0.032	0.005		
Energy use glycerin process heat [MJ kg^{-1} RME]	Normal		1.57	0.079		
Emissions glycerin process heat [kg CO_2eq kg^{-1} RME]	Normal		0.085	0.004		
Energy use glycerin prop. glycol [MJ kg^{-1} RME]	Normal		2.08	0.10		
Emissions glycerin prop. glycol [kg CO_2eq kg^{-1} RME]	Normal		0.22	0.011		
Energy use synt. glyc. prod. [MJ kg^{-1} glycerin]	Normal		164	8.2		
Emissions synt. glyc. prod. [kg CO_2eq kg^{-1} glycerin]	Normal		9.3	0.47		
Energy use crude glyc. refining [MJ kg^{-1} glycerin]	Normal		11.4	0.57		
Emissions crude glyc. refin. [kg CO_2eq kg^{-1} glycerin]	Normal		0.8	0.04		
Market prices [€ ton^{-1}]						
Rapeseed Oil	Normal		500	75		
Rape Meal	Normal		200	20		
RME [3]	Normal		700	105		
Glycerin	Weibull	80			50	2

[1] – Soil carbon stock changes depend on the specific land use change (LUC) scenario considered;
[2] – Depends on the energy and related emissions of each specific industrial process. An uncertainty range of ± 10% for industrial processes is considered, according to Tan et al. (2002);
[3] – RME price is positively correlated with RO price (correlation coef. = 0.8).

4.3.1. Capital Goods

In this dissertation, the energy and emissions linked to capital goods are not taken into account in the life-cycle modeling of biofuels. An estimation of the error associated with this approximation follows, based on the energy embodied in the materials used to construct processing plants and farm machinery. Jungbluth et al. (2007) calculated life-cycle GHG emissions of 88 g CO_2eq GJ^{-1} for a 50-year lifetime oil mill (building and facilities construction, dismantling and elimination/recycling). Acknowledging that GHG emissions from the RME life-cycle[3] amount on average to 46 kg CO_2eq GJ^{-1}, then the emissions associated with the oil mill infrastructure become negligible (0.2%). Concerning farm machinery, Nemecek et al. (2007) calculated 5.80 kg CO_2eq per kg of equipment material (emissions associated with production, maintenance, repair and disposal), for a 3000 kg tractor with a useful life of 7000 hours. Assuming an average tractor fuel consumption of 15 liter per hour (L h^{-1}) and acknowledging that cultivation

[3] This is a conservative figure just for estimation purposes. It is based on the typical GHG emission savings of RME over fossil fuels, with no net carbon emissions from land-use change (EPC 2009). As demonstrated in chapter 6, life-cycle GHG intensity of RME can be much higher, depending on the parameter values and modeling choices considered.

requires on average around 75 L ha^{-1} yr^{-1} of fossil diesel, then 5 h ha^{-1} yr^{-1} of agricultural work are required. Thus, total GHG emissions associated with farm equipment amount to 12.4 kg CO_2eq ha^{-1} yr^{-1} (5 / 7000 x 3000 x 5.8) or 228 g CO_2eq GJ^{-1} (assuming an average energy yield of 1450 kg RME $ha^{-1}yr^{-1}$ x 37.5 MJ kg^{-1} RME = 54.4 GJ $ha^{-1}yr^{-1}$), which represents less than 0.5% of the total RME life-cycle GHG emissions.

Based on this assessment, in which energy is distributed over the total throughput of equipments, it is estimated that the total energy associated with infrastructures represents less than 2%, and thus this contribution is neglected in the analysis. Other biofuel life-cycle studies follow the same approach, as shown in chapter 3. Concerning GHG emissions, this approximation is even more justified, because the GHG intensity of biofuels over the life-cycle includes an additional term: soil GHG emissions, which are not linked to energy inputs. The approach of neglecting capital goods in the life-cycle of RME is also adopted for bioethanol chains (from wheat and sugar beet), due to similarities in terms of equipment and infrastructures used.

Some authors argue that the contribution of agricultural production to the assessment of capital goods is particularly relevant in face of its specific characteristics, namely seasonality and dependence on weather conditions (Frischknecht et al. 2007). The use of machinery in agricultural processes is somehow limited in time, which deviates from typical industrial processes. As a result, the authors claim that the share of capital goods associated to agricultural products should be included in life-cycle studies, particularly in terms of non-renewable energy requirement (Frischknecht et al. 2007).

Nevertheless, in order to avoid idle capacity and reduce machinery ownership costs, subcontracting in the agricultural sector is seen as a competitive advantage (Igata et al. 2008). Smaller farmers cannot maintain modern machinery designed for high volume use, and thus entrust operations to contracting services. In addition to the machinery and technology offered, contractors are chosen for their skills and level of specialization. The UK is the most advanced European country in terms of agricultural contracting systems, but this is a regular practice in other European countries (Igata et al. 2008). One of the consequences of subcontracting is the increase in service time of farming equipment, which then reduces the impact when capital goods are assessed in terms of

an operational functional unit. The lack of detailed information concerning how common are subcontracting practices for agricultural machinery in Europe hinders further quantification on this issue.

4.4. BIOETHANOL AND BioETBE

Bioethanol can be used as motor fuel in (i) pure form; (ii) blended with gasoline; and (iii) after conversion into its derivative bio-ethyl tertiary butyl ether (bioETBE), as shown in Fig. 4.3. The following sections present bioethanol and bioETBE production systems based on major European feedstocks: wheat and sugar beet.

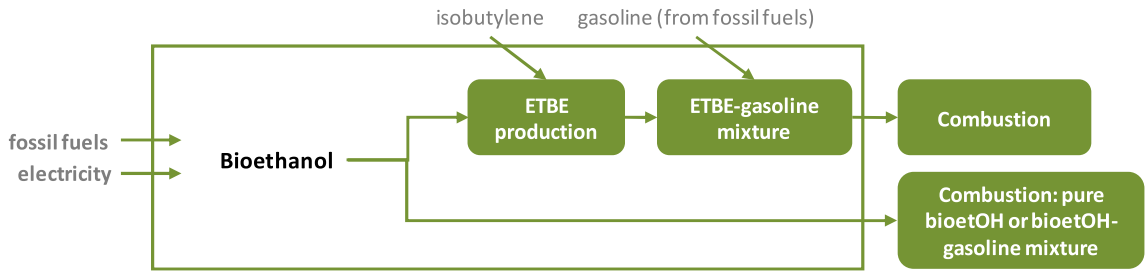


Fig. 4.3. Different uses for bioethanol: (i) as a single fuel; (ii) blended with gasoline; and (iii) blended with gasoline, after conversion in its derivative bioETBE.

4.4.1. Wheat-based Bioethanol

A schematic overview of the production route of wheat-based ethanol is illustrated in Fig. 4.4. Wheat cultivation includes several steps, namely soil preparation (plowing), fertilization, sowing, weed control, and harvesting. Wheat straw is usually plowed back into the field, which has several advantages, as already discussed for rapeseed. After transportation to the processing plant, the raw material is washed to remove any debris, namely soil and stones. The grain goes through a grinding process in a hammer mill, in order to increase the grain surface and maximize the efficiency of the subsequent steps. The milled grain is then mixed with preheated water and liquefaction enzymes, forming a mash and releasing the starch from the cell material. After cooling down, a mixture of amylase enzymes is added to breakdown the starch into simple sugars (saccharification).

Fig. 4.4. Flow chart illustrating the bioethanol production chain from wheat. Transportation activities, potential blending with gasoline and combustion are not shown for the sake of simplicity.

Sugars are fermented to ethanol using yeasts, in a process that yields a solution of 8 to 10% m/m alcohol. This alcohol concentration is increased up to 95-96% v/v by distillation. If ethanol is to be mixed with gasoline for transportation purposes, further dehydration up to 99.7% v/v or higher is required, which is usually achieved through molecular sieves technology. On average, 1 liter of ethanol is produced for each 2.8 kg of wheat processed. After fermentation and distillation, the leftover residue – whole stillage – is pressed and dried to form Distillers Dried Grains with Solubles (DDGS). Approximately 350 kg of DDGS are produced per tonne of wheat. DDGS has high protein and fiber contents and can be sold as feed for ruminants. Börjesson and Tufvesson (2011), for example, estimate that 1 kg of DDGS is equivalent to 0.6 kg of soy meal or 0.4 kg of barley, based on the protein content, whereas Punter et al. (2004) assume that DDGS may displace maize gluten feed or soy meal. Bernesson et al. (2006) consider that DDGS displaces a mix of soy bean meal and soy oil with equivalent protein and energy contents.

Additional details concerning the production of bioethanol from wheat can be found e.g. in Punter et al. (2004), SenterNovem (2006), Smith et al. (2006), Ingledew (2009), ADEME (2010) and Walker (2010). Table 4.5 gathers data collated from several sources to build the life-cycle inventory of wheat-based bioethanol. Parameter uncertainty has been addressed using the probability density functions listed in Table 4.6.

Table 4.5. Parameter values considered in the life-cycle inventory of wheat-based bioethanol.

Parameter [(1)]				**Source**
Seed rate	*Min*	*Med*	*Max*	
[kg ha^{-1} yr^{-1}]		185		*Richards (2000)*
		120		*Elsayed et al. (2003)*
		120		*Mortimer et al. (2004)*
		135		*ADEME (2010)*
		148		*Elsgaard (2010)*
		230		*Marakoglu & Çarman (2010)*
N fertilizer application rate	*Min*	*Med*	*Max*	
[kg ha^{-1} yr^{-1}]		183		*Richards (2000)*
		185		*Mortimer et al. (2004)*
	146		198	*Swanston & Newton (2005)*
		120		*Bernesson et al. (2006)*
	100		235	*Wangstrand et al. (2007)*
	80		160	*Popa et al. (2008)*
	0		240	*Kindred et al. (2008)*
		186		*ADEME (2010)*
		191		*de Vries et al. (2010)*
	146		174	*Elsgaard (2010)*
	65		183	*Fertistat (2010)*
		150		*Glubiak&Korzeniowska (2010)*
		150		*Börjesson&Tufvesson (2011)*
P_2O_5 fertilizer application rate		68		*Richards (2000)*
[kg ha^{-1} yr^{-1}]		60		*Elsayed et al. (2003)*
		30.2		*Bernesson et al. (2006)*
		124.2		*Popa et al. (2008)*
		28		*ADEME (2010)*
		39.1		*Elsgaard (2010)*
	40.8		142	*Fertistat (2010)*
		30.2		*Glubiak&Korzeniowska (2010)*
		44.3		*Börjesson & Tufvesson (2011)*
K_2O fertilizer application rate		77		*Richards (2000)*
[kg ha^{-1} yr^{-1}]		60		*Elsayed et al. (2003)*
		34.9		*Bernesson et al. (2006)*
		26		*ADEME (2010)*
		79.5		*Elsgaard (2010)*
	34.9		84.3	*Fertistat (2010)*
		60.2		*Glubiak&Korzeniowska (2010)*
		12		*Börjesson &Tufvesson (2011)*
Pesticides application rate		5.50		*Richards (2000)*
[kg ha^{-1} yr^{-1}]		8.41		*Elsayed et al. (2003)*
		4.80		*de Vries et al. (2010)*
		1.68		*Elsgaard (2010)*
Fuel consumption of agricultural machinery (diesel)		105.6		*Elsayed et al. (2003)*
[liter ha^{-1}]		139		*Punter et al. (2004)*
		81 – 93		*Dalgaard & Dalgaard (2006)*
		97		*Hvid (2009)*
		100		*ADEME (2010)*
		52.5 ± 10%		*Elsgaard (2010)*
N_2O emissions from soil		*default*	*range*	
[kg N_2O kg^{-1} N fert]				
			0.007 – 0.041	*Kaiser et al. (1998)*
		0.01	0.004 – 0.017	*Richards (2000)*

				Freibauer &Kaltschmitt
		1.6 – 7.8 kg N_2O ha^{-1}		*(2001)*
		0.005		*ADEME (2002)*
		0.024		*Punter et al. (2004)*
			0.016 – 0.035	*SenterNovem (2005a)*
			0.3 – 0.9 kg N_2O ha^{-1}	*Tzilivakis et al. (2005)*
		0.0208	0.00489 – 0.102	*IPCC (2006)*
		2.23 kg N_2O ha^{-1}	0.74–3.72 kg N_2O ha^{-1}	*JEC (2007)*
		0.047-0.079		*Crutzen et al. (2008)*
		0.01		*Lechón et al. (2009)*
		0.0255	0.004 – 0.112	*Soimakallio et al. (2009)*
		4.17 kg N_2O ha^{-1}		*FA (2010)*
		0.01 – 0.0175		*GA (2010)*
		0.024		*Börjesson &Tufvesson (2011)*
Agricultural yield	*Min*	*Med*	*Max*	
[t ha^{-1}]		8.0		*Herbert (1995)*
		9.0		*ADEME (2002)*
		6.9		*Elsayed et al. (2003)*
		8.6		*Mortimer et al. (2004)*
		6.9		*Punter et al. (2004)*
		6.3		*Bernesson et al. (2006)*
		7.5		*Börjesson (2009)*
		6.4		*Gnansounou et al. (2009)*
		7.8		*ADEME (2010)*
		8.2		*de Vries et al. (2010)*
	3.2	3.6 – 3.9	4.2	*FAOSTAT (2010)* [(2)]
	6.2	6.9 – 7.9	8.3	*FAOSTAT (2010)* [(3)]
	4.2	6.4	8.6	*Börjesson &Tufvesson (2011)*
Transportation activities				
Distances traveled, single trip [km]				
From farms to processing facilities		130		*Elsayed et al. (2003)*
		50		*Gnansounou et al. (2009)*
From plants to local distribution depots		200		*Poitrat et al. (1998)*
		225		*Elsayed et al. (2003)*
		250		*Gnansounou et al. (2009)*
Bioethanol production (conversion ratios)				
Bioethanol		*t etOH t^{-1} wheat*		
		0.279		*Herbert (1995)*
		0.279		*Rozakis et al. (2001)*
		0.283		*ADEME (2002)*
		0.330		*Elsayed et al. (2003)*
		0.330		*Mortimer et al. (2004)*
		0.330		*Punter et al. (2004)*
		0.299		*Gnansounou et al. (2009)*
		0.283 – 0.309		*ADEME (2010)*
		0.281		*de Vries et al. (2010)*
		0.294		*ePURE (2010)*
		0.378		*Börjesson&Tufvesson (2011)*
Distillers Dried Grains with Solubles (DDGS)		*t DDGS t^{-1} wheat*		
		0.418		*Herbert (1995)*
		0.351		*Rozakis et al. (2001)*
		0.497		*Elsayed et al. (2003)*
		0.497		*Mortimer et al. (2004)*
		0.376		*Punter et al. (2004)*
		0.363		*Gnansounou et al. (2009)*
		0.381		*Börjesson&Tufvesson (2011)*
		0.370		*ePURE (2010)*

Energy use				
Fuel oil (wheat drying) [MJ t^{-1} wheat]		661		*Elsayed et al. (2003)*
Electricity (wheat drying) [MJ t^{-1} wheat]		42		*Punter et al. (2004)*
Diesel (wheat drying) [MJ t^{-1} wheat]		660		*Punter et al. (2004)*
Electricity (wheat drying) [MJ t^{-1} wheat]		44.2		*JEC (2007)*
Diesel (wheat drying) [MJ t^{-1} wheat]		680		*JEC (2007)*
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.0096		*Elsayed et al. (2003)*
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.054		*Punter et al. (2004)*
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.054		*JEC (2007)*
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.012 – 0.015		*ADEME (2010)*
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]		0.241		*Elsayed et al. (2003)*
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]		0.204		*Punter et al. (2004)*
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]		0.404		*JEC (2007)* [(4)]
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]		0.28 – 0.33		*ADEME (2010)*
DDGS drying [MJ MJ^{-1} etOH]		0.196		*Elsayed et al. (2003)*
DDGS drying [MJ MJ^{-1} etOH]		0.160		*Punter et al. (2004)*
DDGS drying [MJ MJ^{-1} etOH]		0.160		*ADEME (2010)*
Co-product substitution ratio		*kg soy meal kg^{-1} DDGS*		
DDGS vs. Soy meal		0.78		*JEC (2007)*
		0.82		*Gnansounou et al. (2009)*
		0.60		*Börjesson & Tufvesson (2011)*
Substitution credits [(6)]		*MJ kg^{-1} soy meal*	*$kgCO_2eq$ kg^{-1} soymeal*	
DDGS vs. Soy meal				
Market prices [€ t^{-1}]	*Min*	*Med*	*Max*	
Bioethanol		675		*Bernesson et al. (2006)*
		887		*Gnansounou et al. (2009)*
	579	781	869	*Börjesson &Tufvesson (2011)*
DDGS	4.5		108	*Bernesson et al. (2006)*
		195		*Gnansounou et al. (2009)*
	130	180	210	*Börjesson &Tufvesson (2011)*

[(1)] – Energy and GHG emissions of fertilizer and pesticides production, as well as generic data for biofuel life-cycle modeling are provided in appendix (Table A.1);
[(2)] – Ranges for low productivity wheat producers;
[(3)] – Ranges for high productivity wheat producers;
[(4)] – Includes DDGS drying;
[(5)] – Ranges are due to different processes in ETBE manufacturing: one process is based on isobutylene from refining and the other is a co-product valorization from propylene oxide manufacturing; it was not possible however to split the data between processes;
[(6)] – The credits for soy meal substitution are listed in Table 4.3;

Table 4.6. Wheat-based bioethanol: Probability distributions for input data and characteristic values computed for each distribution.

Parameter	distribution	min	mean	max	std dev
Fertilizer application rates [kg ha^{-1} yr^{-1}]					
N fertilizer	Normal		160		20
P_2O_5 fertilizer	Lognormal		40		8
K_2O fertilizer	Normal		60		12
Seed application rate [kg ha^{-1} yr^{-1}]	Normal		160		20
Pesticides application rate [kg ha^{-1} yr^{-1}]	Normal		5.0		1.0
Fuel consumption agricultural machinery [l ha^{-1}]	Normal		80		16
Soil carbon stock changes [ton C ha^{-1} yr^{-1}] [(1)]					
N_2O emissions from soil [kg N_2O kg^{-1} N fert]	Lognormal		0.0208		0.021
Agricultural yield [ton ha^{-1}]	Normal		7.4		0.74
Ratio ethanol / wheat [kg kg^{-1}]	Uniform	0.29		0.32	
Ratio DDGS / wheat [kg kg^{-1}]	Uniform	0.36		0.42	
Wheat transport [km]	Normal		60		12

Ethanol transport [km]	Normal	175	17.5
Energy use in generic industrial processes [2]	Normal	[3]	Mean/20
Electricity (grain drying) [MJ kg^{-1} wheat]	Normal	0.04	0.002
Natural Gas (grain drying) [MJ kg^{-1} wheat]	Normal	0.3	0.015
Electricity (etOH prod.) [MJ MJ^{-1} etOH]	Normal	0.0135	0.00068
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]	Normal	0.24	0.012
Natural Gas (DDGS drying) [MJ MJ^{-1} etOH]	Normal	0.16	0.008
Substitution credits			
Ratio soy meal / DDGS [kg kg^{-1}]	Normal	0.76	0.038
Energy use in soy meal production [MJ kg^{-1}]	Normal	3.53	0.29
Emissions in soy meal production [kg CO_2eq kg^{-1}]	Normal	0.065	0.097
Market prices [€ ton^{-1}]			
Ethanol	Normal	750	75
DDGS	Normal	180	18

[1] – Soil carbon stock changes depend on the specific land use change (LUC) scenario considered;
[2] – Depends on the energy and related emissions of each specific industrial process.

4.4.2. Sugar beet-based Bioethanol

Sugar beet cultivation allows a wide variety of soil and climate conditions. For this reason, the crop is produced in most European countries. Main EU sugar beet producers in 2009 were France, Germany, Poland, the United Kingdom and the Netherlands. Sugar beet cultivation includes several steps, namely soil preparation (plowing), fertilizer application, sowing, weed control, and harvesting. In addition to the crop (beets), sugar beet tops are also produced, and remain in the field for grazing purposes, especially for sheep but also for cattle. After transportation to a processing plant, the beets are washed and sliced into chips. Slicing maximizes the efficiency of next step –diffusion–, in which the chips are passed into a hot water solution to extract the sweet/raw juice. Beet pulps are the most important co-product of beet processing. On average, 75 kg of dried beet pulps are produced per tonne of sugar beet processed. After pressing and drying, pulps can be sold as animal-feed concentrate displacing equivalent products from cereal fermentation (low-protein animal feed), or can be burnt for process heat. Usually, sugar factories prefer the feed pathway, because pulps worth more as feed than fuel (JEC 2007). Examples of food products displaced by sugar beet pulps in animal feed include wheat grain (JEC 2007; Halleux et al. 2008), barley (Renouf et al. 2008; Börjesson and Tufvesson 2011), and protein peas (Halleux et al. 2008). Substitution ratios are calculated on the basis of energy, protein, sugar and/or starch content.

The raw juice exiting the diffuser can be used for bioethanol or sugar production (Fig. 4.5). In the former, the sugar juice is fermented to a low concentration ethanol solution (8-10% m/m) using yeast, usually Saccharomyces cerevisiae. Ethanol concentration is then increased up to 95-96% v/v in a distillation process, yielding hydrous ethanol. At this concentration level, ethanol and water form an azeotrope, meaning that the mixture composition cannot be further changed by distillation. A final step is required to obtain anhydrous ethanol (99.7% v/v or higher), such as the use of molecular sieve adsorption technology. Each tonne of sugar beet processed, at 16% sugar content, yields ca. 100 liters of ethanol. Vinasses (stillage) is a by-product from ethanol distillation and, after concentration, can be sold as an additive for animal feed or can be spread on agricultural land (fertilizer). Per liter of bioethanol produced, nearly 0.6 kg of concentrated vinasses is obtained.

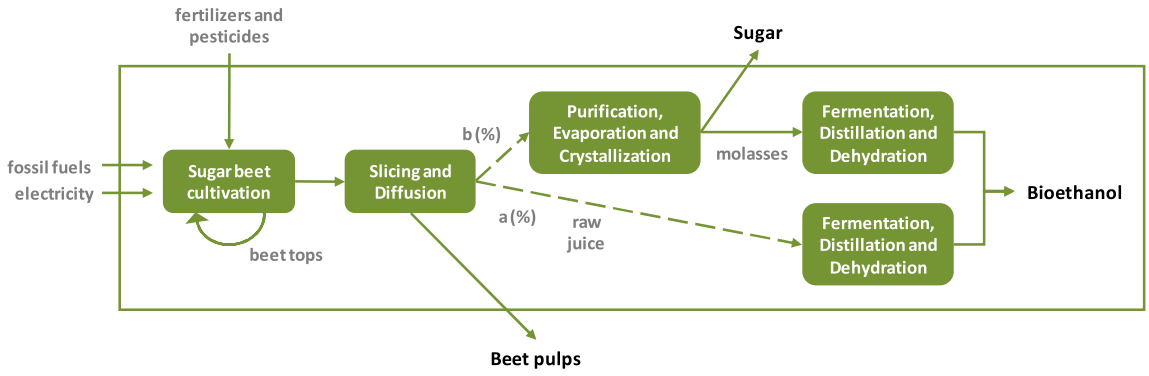


Fig. 4.5. Flow chart illustrating the bioethanol production chain from sugar beet. Combined production of bioethanol and sugar is possible, depending on the market conditions (dashed lines). Transportation activities, potential blending with gasoline, and combustion are not shown for the sake of simplicity.

The technological processes involved to obtain ethanol from sugar beet may not be self-dedicated to the production of ethanol. Instead, the whole chain may be shared by the alcohol and sugar industries. When sugar production is also envisaged (sugar pathway), the raw juice is firstly purified and partially evaporated, yielding syrup that is then

crystallized to sugar. The co-product of sugar crystallization –molasses– can be fermented to ethanol in a process similar to the one described above for sugar juice. Depending on the agricultural productivity and the sugar content of the beets, sugar yields can reach 10 tonnes per hectare of cultivated crop.

The trade-off between sugar and bioethanol returns is a key factor driving the allocation of sugar beet between sugar and bioethanol production. Depending on the market conditions, the sugar beet transformation industry may pursue the sugar and/or bioethanol options (Jolly 2003; Krajnc and Glavic 2009). Recently, following the European sugar production reform in which the economic support for refined sugar was reduced owing to the surplus of sugar on the world market, many sugar producers have considered shifting to bioethanol production (Krajnc and Glavic 2009). Both the bioethanol and sugar pathways from sugar beet are addressed in this dissertation.

In contrast to joint production, in which the relative output volume of the co-products is fixed, the share of sugar and ethanol extracted from sugar beet is independently variable (combined production, illustrated in Fig. 4.5 by dashed lines), which offers the sugar beet transformation industry the opportunity to broaden its revenue base and to assure continued financial viability by pursuing the ethanol and/or sugar options. For combined production, allocation can be avoided simply by modeling directly the consequences of a change in the output of the co-product of interest (that which is used in the product system under study) (Weidema 2001, 2003).

Details concerning the technological description and the mass and energy balances of bioethanol production from sugar beet can be found e.g. in Poitrat et al. (1998), Tzilivakis et al. (2005), Draycott (2006), Malça and Freire (2006a), Asadi (2007), and ADEME (2010). Table 4.7 lists data collected from several sources concerning the parameters used in the life-cycle modeling of sugar beet-based bioethanol. Table 4.8 shows the selected probability density functions through which parameter uncertainty is incorporated in the model.

Table 4.7. Parameter values considered for the life-cycle inventory of sugar beet-based bioethanol.

Parameter [(1)]				**Source**
Seed rate	*Min*	*Med*	*Max*	
[kg ha^{-1} yr^{-1}]		4		*Elsayed et al. (2003)*
N fertilizer application rate	*Min*	*Med*	*Max*	
[kg ha^{-1} yr^{-1}]		142.5		*Poitrat et al. (1998)*
		147.4		*Elsayed et al. (2003)*
		103		*Mortimer et al. (2004)*
		135		*Halleux et al. (2008)*
	110	112	120	*Renouf et al. (2008)*
		128		*ADEME (2010)*
		107		*de Vries et al. (2010)*
	100		145	*Fertistat (2010)*
	100		150	*Monti (2010)*
		120		*Börjesson&Tufvesson (2011)*
P_2O_5 fertilizer application rate		70		*Poitrat et al. (1998)*
[kg ha^{-1} yr^{-1}]		56		*Elsayed et al. (2003)*
		125		*Halleux et al. (2008)*
	31.9		39.0	*Renouf et al. (2008)*
		47.8		*ADEME (2010)*
	67.4		124.2	*Fertistat (2010)*
		35.5		*Börjesson&Tufvesson (2011)*
K_2O fertilizer application rate		175		*Poitrat et al. (1998)*
[kg ha^{-1} yr^{-1}]		141		*Elsayed et al. (2003)*
		212.5		*Halleux et al. (2008)*
	49.4	61.4	100.0	*Renouf et al. (2008)*
		159.3		*ADEME (2010)*
	42.1		186.7	*Fertistat (2010)*
		48.2		*Börjesson&Tufvesson (2011)*
Pesticides application rate		4.37		*Poitrat et al. (1998)*
[kg ha^{-1} yr^{-1}]		1.15		*Elsayed et al. (2003)*
		5.30		*Halleux et al. (2008)*
	4.2	8.6	24.1	*Renouf et al. (2008)*
		3.5		*de Vries et al. (2010)*
Fuel consumption of agricultural machinery (diesel)		161		*Poitrat et al. (1998)*
[liter ha^{-1}]		83.3		*Elsayed et al. (2003)*
		119.5		*Halleux et al. (2008)*
	188.4	238.1	292.4	*Renouf et al. (2008)*
		170		*ADEME (2010)*
N_2O emissions from soil		*default*	*range*	
[kg N_2O kg^{-1} N fert]			0.007 – 0.041	*Kaiser et al. (1998)*
		0.0021		*Poitrat et al. (1998)*
		1.6 – 7.8 kg N_2O ha^{-1}		*Freibauer &Kaltschmitt (2001)*
		0.005		*ADEME (2002)*
		0.025	0.0043 – 0.060	*GM (2002)*
			0.016 – 0.035	*SenterNovem (2005)*
			0.5 – 2.0 kg N_2O ha^{-1}	*Tzilivakis et al. (2005)*
		0.0208	0.00489 – 0.102	*IPCC (2006)*
		2.79 kg N_2O ha^{-1}	1.91–3.67 kg N_2O ha^{-1}	*JEC (2007)*
		0.047-0.079		*Crutzen et al. (2008)*
		0.063		*Renouf et al. (2008)*
		0.0255	0.004 – 0.112	*Soimakallio et al. (2009)*

		0.022	± 50%	*ADEME (2010)*
		4.68 kg N_2O ha^{-1}		*FA (2010)*
		0.01 – 0.0175		*GA (2010)*
		0.025		*Börjesson&Tufvesson (2011)*
Agricultural yield	*Min*	*Med*	*Max*	
[t ha^{-1}]		67.3		*Poitrat et al. (1998)*
		66.2		*ADEME (2002)*
		56.4		*Elsayed et al. (2003)*
	47.8	49.6	57.3	*Renouf et al. (2008)*
		80		*ADEME (2010)* [(2)]
		61.2		*de Vries et al. (2010)*
	42	48 – 70	79	*FAOSTAT (2010)* [(3)]
	58	62 – 85	94	*FAOSTAT (2010)* [(4)]
Transportation activities				
Distances traveled, single trip [km]				
From farms to processing facilities		25		*Poitrat et al. (1998)*
		40		*Elsayed et al. (2003)*
		47		*Mortimer et al. (2004)*
		46		*Renouf et al. (2008)*
		50		*Börjesson&Tufvesson (2011)*
From plants to local distribution depots		200		*Poitrat et al. (1998)*
		225		*Elsayed et al. (2003)*
Bioethanol production (conversion ratios)				
Bioethanol				
[t etOH t^{-1} sugar beet]		0.086		*Poitrat et al. (1998)*
		0.017 (sugar pathway)		*Poitrat et al. (1998)*
		0.087		*ADEME (2002)*
		0.101		*ePURE (2010)*
Sugar beet pulps				
[t pulps t^{-1} sugar beet]		0.052		*Poitrat et al. (1998)*
		0.076		*ePURE (2010)*
Sugar beet pulps				
[t pulps t^{-1} etOH]		0.746		*Halleux et al. (2008)*
		0.856		*Börjesson&Tufvesson (2011)*
Sugar				
[t sugar t^{-1} sugar beet]		0.178		*Poitrat et al. (1998)*
		0.177		*ADEME (2010)*
		0.14 – 0.20		*Monti (2010)*
Bioethanol production (energy use)				
Bioethanol pathway				
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.024		*Elsayed et al. (2003)*
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.0172		*JEC (2007)*
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.044		*Halleux et al. (2008)*
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.020 – 0.028		*ADEME (2010)*
Energy (etOH prod.) [MJ MJ^{-1} etOH]		0.487		*Poitrat et al. (1998)* [(5)]
Fuel Oil (etOH prod.) [MJ MJ^{-1} etOH]		0.320		*Elsayed et al. (2003)*
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]		0.304		*JEC (2007)*
Heat (etOH prod.) [MJ MJ^{-1} etOH]		0.233		*Halleux et al. (2008)*
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]		0.30 – 0.35		*ADEME (2010)*
Electricity (pulps drying) [MJ MJ^{-1} etOH]		0.018		*JEC (2007)*
Electricity (pulps drying) [MJ MJ^{-1} etOH]		0.018		*Halleux et al. (2008)*
Natural Gas (pulps drying) [MJ MJ^{-1} etOH]		0.236		*JEC (2007)*
Heat (pulps drying) [MJ MJ^{-1} etOH]		0.213		*Halleux et al. (2008)*
Sugar pathway				
Electricity (etOH prod.) [MJ MJ^{-1} etOH]		0.041		*Poitrat et al. (1998)*
Energy (etOH prod.) [MJ MJ^{-1} etOH]		2.141		*Poitrat et al. (1998)* [(5)]
Co-product substitution ratio		*kg wheat kg^{-1} pulps*		
Pulps vs. Wheat		0.83		*JEC (2007)*
Substitution credits				
Pulps vs. Wheat		*MJ kg^{-1} wheat*		*$kgCO_2eq$ kg^{-1} wheat*

		2.22	0.31	*JEC (2007)*
Sugar vs. Imported sugar [6]		*MJ kg^{-1} imp. sugar*	*kgCO$_2$eq kg^{-1} imp.sugar*	
Sugarcane cultivation (Morocco)		3.76 – 8.18	n/a	*Mrini et al. (2001)*
Sugarcane cultivation (Brazil)		1.64	0.151	*Jungbluth et al. (2007)*
Sugarcane cultivation (Brazil)		1.66	0.290	*Macedo et al. (2008)*
Sugarcane cultivation (South Africa)		3.41	0.417	*Mashoko et al. (2010)*
Sugar production (Brazil)		0.49	0.039	*Jungbluth et al. (2007)*
Sugar production (South Africa)		1.48	0.108	*Mashoko et al. (2010)*
Sugar transport to the EU [7]		2.0	0.137	*Jungbluth et al. (2007)*
Market prices [€ t^{-1}]	*Min*	*Med*	*Max*	
Bioethanol		675		*Bernesson et al. (2006)*
		887		*Gnansounou et al. (2009)*
	579	781	869	*Börjesson&Tufvesson (2011)*
Sugar beet pulps				
	120	170	190	*Börjesson&Tufvesson (2011)*
Sugar				
	277		435	*GAIN (2010)*
	160		287	*FAOSTAT (2010)*

[1] – Energy and GHG emissions of fertilizer and pesticides production, as well as generic data for biofuel life-cycle modeling are provided in appendix (Table A.1);
[2] – Soiled sugar beet (includes soil, stones, dirt);
[3] – Ranges for low productivity sugar beet producers;
[4] – Ranges for high productivity sugar beet producers;
[5] – Different energy sources were used: fuel oil, coal and gas;
[6] – Sugar imports to the EU have been assumed for estimation of energy and GHG emission credits in the sugar beet-based bioethanol chain (sugar pathway);
[7] – Transport from Brazil to the EU (includes several transportation modes: rail, lorry and transoceanic vessel).

Table 4.8. Sugar beet-based bioethanol: Probability distributions for input data and characteristic values computed for each distribution.

Parameter	distribution	mean	std dev
Fertilizer application rates [kg ha^{-1} yr^{-1}]			
N fertilizer	Normal	125	12.5
P_2O_5 fertilizer	Normal	85	12.8
K_2O fertilizer	Normal	160	16
Pesticides application rate [kg ha^{-1} yr^{-1}]	Normal	3.6	0.54
Fuel consumption agricultural machinery [l ha^{-1}]	Normal	120	12
Soil carbon stock changes [ton C ha^{-1} yr^{-1}] [1]			
N_2O emissions from soil [kg N_2O kg^{-1} N fert]	Lognormal	0.0208	0.021
Agricultural yield [ton ha^{-1}]	Normal	73.6	7.36
Ratio pulps / sugar beet [kg kg^{-1}]	Normal	0.06	0.01
Ratio sugar beet / ethanol (via etOH) [kg kg^{-1}]	Normal	10.8	0.54
Ratio sugar beet / ethanol (via sugar) [kg kg^{-1}]	Normal	58.5	2.9
Ratio sugar /sugar beet [kg kg^{-1}]	Normal	0.14	0.01
Sugar beet transport [km]	Normal	40	6
Ethanol transport [km]	Normal	175	17.5
Energy use in generic industrial processes [2]	Normal	[3]	Mean/20
Bioethanol pathway			
Electricity (etOH prod.) [MJ MJ^{-1} etOH]	Normal	0.022	0.0011
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]	Normal	0.30	0.015
Sugar pathway			
Electricity (etOH prod.) [MJ MJ^{-1} etOH]	Normal	0.16	0.008
Natural Gas (etOH prod.) [MJ MJ^{-1} etOH]	Normal	1.14	0.057
Substitution credits [3]			
Energy use in wheat production [MJ kg^{-1}]	Normal	0.13	0.007
Emissions in wheat production [kg CO_2eq kg^{-1}]	Normal	0.0183	0.0028

Energy, imported sugar [MJ kg^{-1}]	Normal	4.5	0.225
Emissions, imported sugar [kg CO_2eq kg^{-1}]	Normal	0.4	0.04
Market prices [€ ton^{-1}]			
Ethanol	Normal	750	75
Pulps [4]	Normal	150	15
Sugar	Normal	300	50

[1] – Soil carbon stock changes depend on the specific land use change (LUC) scenario considered;
[2] – Depends on the energy and related emissions of each specific industrial process;
[3] – Market price for pulps is estimated from wheat prices, which ranged from 144 to 252 € t^{-1} in the period 2006-2010 (FAOSTAT 2010). It is assumed that sugar beet pulps replace wheat grain in animal feed (1 MJ of sugar beet pulps replaces 0.83 MJ of wheat grain, JEC 2007), based on the protein content and digestible energy content of these alternative options [JEC 2007, WTT appendix 1, p.48]. Data from Börjesson and Tufvesson (2011) has also been taken into account in the estimation of pulps price.

4.4.3. BioETBE

Bioethanol can be converted into its derivative bio-ethyl tertiary butyl ether (bioETBE) through chemical reaction with isobutylene at the petroleum refinery (Fig. 4.6). BioETBE offers the same benefits as bioethanol (e.g. improved combustion, increased fuel octane, reduced oil imports) without the technical and logistic difficulties shown by the alcohol, namely (i) storage and shipping challenges to avoid water contamination; (ii) much lower volumetric energy content than gasoline; and (iii) increased evaporative emissions due to higher volatility of bioethanol/gasoline blends (Bensaid 2004; His 2004). BioETBE can be used at rates up to 22% v/v in standard spark ignition engines without technical modification. France and Spain are examples in which bioethanol production is partially converted into bioETBE, prior to blending with gasoline.

BioETBE contributes partially to the share of renewable fuels in the transportation sector, as the percentage of bioETBE obtained from bioethanol amounts to 47% m/m. The remaining 53% m/m comes from isobutylene, a by-product of the petroleum refining process (i.e. fossil), which requires high energy inputs in the synthesis process. Following a similar approach to that used in the combustion of biodiesel (RME), tailpipe CO_2 emissions from bioETBE combustion are considered as not completely neutral, due to the non-renewable nature of the isobutylene feedstock used for bioETBE synthesis. This fraction of bioETBE contributes to net CO_2 emissions during combustion (48 g CO_2eq MJ^{-1}, according to JEC 2007), and must be factored in.

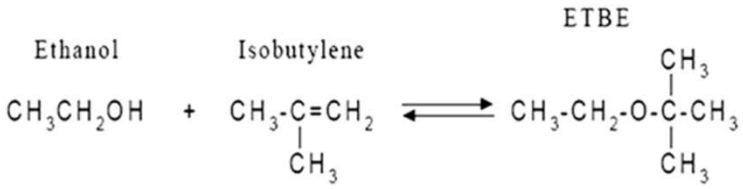


Fig. 4.6. (Bio)ETBE is produced at the petroleum refinery through the chemical reaction of (bio)ethanol with (fossil) isobutylene, in the presence of heat and a catalyst.

Table 4.9 lists literature data for the synthesis of bioETBE and the parameter values considered in this dissertation.

Table 4.9. Parameter values considered for the bioETBE synthesis process.

Parameter			**Source**
BioETBE synthesis (feedstock)			
Bioethanol (ETBE synt.) [t t^{-1} ETBE]	0.47		*Poitrat et al. (1998); ADEME (2010)*
Isobutylene (ETBE synt.) [t t^{-1} ETBE]	0.53		*Poitrat et al. (1998); ADEME (2010)*
BioETBE synthesis (energy use)			
Electricity (ETBE synt.) [MJ t^{-1} ETBE]	50		*Poitrat et al. (1998)*
Electricity (ETBE synt.) [MJ t^{-1} ETBE]	200 – 500		*ADEME (2010)* [5]
Natural Gas (ETBE synt.) [MJ t^{-1} ETBE]	2700		*Poitrat et al. (1998)*
Natural Gas (ETBE synt.) [MJ t^{-1} ETBE]	2000 – 3000		*ADEME (2010)* [5]
Parameter	**distribution**	**mean**	**std dev**
Electricity (ETBE synt.) [MJ t^{-1} ETBE]	Normal	250	6.25
Natural Gas (ETBE synt.) [MJ t^{-1} ETBE]	Normal	2700	67.5

4.5. SENSITIVITY ANALYSIS

A preliminary sensitivity analysis has been performed to evaluate the response of each biofuel model to variations in individual parameters. The contribution of each parameter to the overall uncertainty of results is a combination of two factors: (i) the model sensitivity to the parameter; and (ii) the inherent uncertainty of the parameter. In particular, parameters for which the literature showed large ranges of variation have been selected. Low and high estimates for the selected set of parameters based on realistic ranges from literature data are shown in Tables 4.10, 4.11 and 4.12, respectively

for RO/RME, bioethanol and bioETBE from wheat, and bioethanol and bioETBE from sugar beet.

Table 4.10. Rapeseed Oil and Rapeseed Methyl Ester: selected parameters and ranges for sensitivity analysis.

Parameter	Units	Low	High
Rapeseed yield	ton ha^{-1}	3.0	4.0
N fertilizer application rate	kg N ha^{-1}	120	180
N fertilizer production	MJ kg^{-1} N	50	60
N fertilizer production	kg CO_2eq kg^{-1} N	4.0	8.0
Diesel fuel agric. machinery	L ha^{-1}	50	100
Soil N_2O emissions	g N_2O kg^{-1} N	5	40
Soil carbon emissions	t C ha^{-1} yr^{-1}	−0.4	0.4
Oil extraction rate	%	39	43
Soy meal / rape meal ratio	kg kg^{-1}	0.74	0.82
Soy meal prod. energy credit	MJ kg^{-1} soy meal	3.2	3.8
Soy meal prod. GHG credit	g CO_2eq kg^{-1} soy meal	0	150
N_2O GWP	kg CO_2eq kg^{-1} N_2O	250	350

Table 4.11. Wheat bioethanol/bioETBE: selected parameters and ranges for sensitivity analysis.

Parameter	Units	Low	High
Wheat yield	ton ha^{-1}	6.4	8.4
N fertilizer application rate	kg N ha^{-1}	140	180
N fertilizer production	MJ kg^{-1} N	50	60
N fertilizer production	kg CO_2eq kg^{-1} N	4.0	8.0
Diesel fuel agric. machinery	L ha^{-1}	60	100
Soil N_2O emissions	g N_2O kg^{-1} N	5	40
Soil carbon emissions	t C ha^{-1} yr^{-1}	−0.4	0.4
DDGS / soy meal ratio	kg soy meal kg^{-1} DDGS	0.72	0.80
Bioethanol production	MJ MJ^{-1} etOH	0.29	0.32
Soy meal prod. energy credit	MJ kg^{-1} soy meal	3.2	3.8
Soy meal prod. GHG credit	g CO_2eq kg^{-1} soy meal	0	150
Isobutylene production	MJ kg^{-1} isobutylene	52	53
N_2O GWP	kg CO_2eq kg^{-1} N_2O	250	350

Table 4.12. Sugar beet bioethanol/bioETBE: selected parameters and ranges for sensitivity analysis.

Parameter	Units	Low	High
Sugar beet yield	ton ha^{-1}	60	80
N fertilizer application rate	kg N ha^{-1}	100	150
N fertilizer production	kg CO_2eq kg^{-1} N	4.0	8.0
Diesel fuel agric. machinery	L ha^{-1}	100	140
Soil N_2O emissions	g N_2O kg^{-1} N	5	40
Soil carbon emissions	t C ha^{-1} yr^{-1}	−0.4	0.4
Sugar beet / bioethanol ratio	kg sugar beet kg^{-1} etOH	10.0	11.6
Raw juice production	MJ t^{-1} sugar beet	170	190
Bioethanol production	MJ MJ^{-1} etOH	0.38	0.42
Pulps / sugar beet ratio	kg kg^{-1}	0.055	0.070
Wheat prod. GHG credit	g CO_2eq kg^{-1} wheat	200	425
Isobutylene production	MJ kg^{-1} isobutylene	52	53
Energy ETBE synthesis	MJ kg^{-1} ETBE	2.6	2.85

The model sensitivity to each individual parameter is illustrated by the slopes presented in Figs. 4.7 to 4.18. The variation range for each parameter has been computed using the 1st and 99th percentiles of the intervals listed in Tables 4.10 to 4.12[4]. Several parameters affect the energy renewability efficiency and GHG intensity of the investigated biofuel systems, either in the increasing (positive slope) or decreasing (negative slope) directions. Although some graphs in Figs. 4.7 to 4.18 are very similar, they are shown to illustrate the varying magnitudes of ERenEf and GHG intensity between biofuel chains.

Concerning ERenEf results (Figs. 4.7 to 4.12), several parameters affect the energy balance. Most important are the agricultural yield and nitrogen fertilizer application rate (rapeseed and wheat chains), and isobutylene production (wheat bioETBE). The energy renewability of bioethanol/bioETBE from sugar beet (bioethanol pathway) is almost equally influenced by several parameters, namely input/output ratios at the technological level (amount of pulps and bioethanol produced per kg of sugar beet processed), and energy inputs for bioethanol and isobutylene (bioETBE chain) production.

[4] The software used to perform the preliminary sensitivity analysis only allowed fixed testing ranges.

In terms of GHG intensity (Figs. 4.13 to 4.18), soil carbon emissions are the most important parameter for the analyzed biofuel systems, but life-cycle GHG emissions are also sensitive to soil N_2O emissions. Other parameters like agricultural yield, N fertilizer application rate, GHG credits of co-products displacing other products, and global warming potential (GWP) of nitrous oxide also influence the GHG balance, although to a lesser extent.

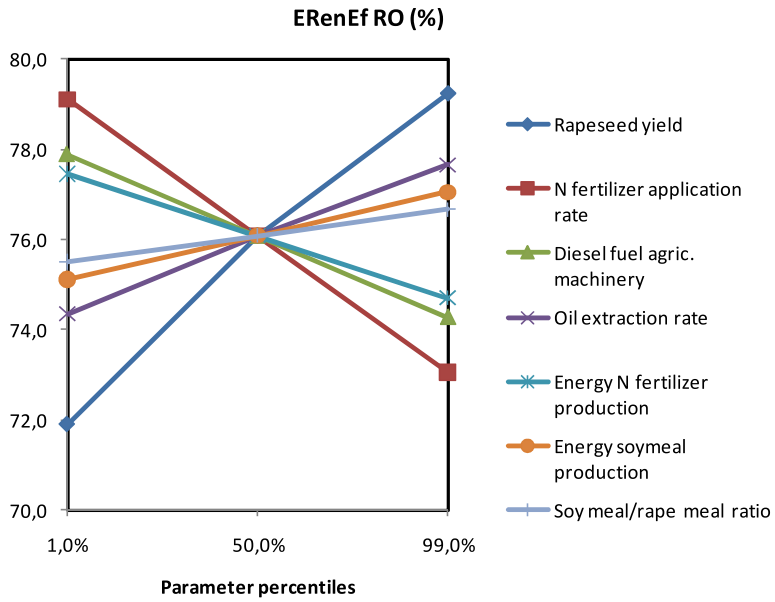


Fig. 4.7. Rapeseed Oil (subst. method): sensitivity of ERenEf to selected parameters.

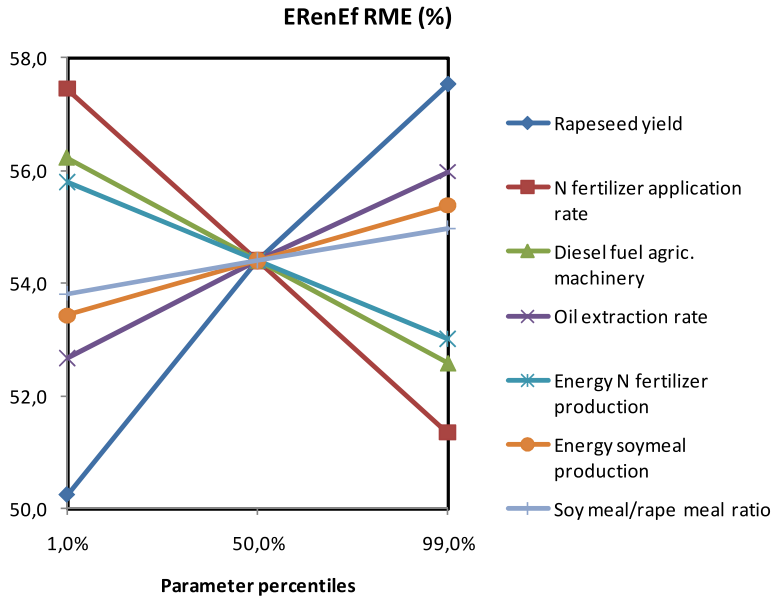


Fig. 4.8. Rapeseed Methyl Ester (subst. method): sensitivity of ERenEf to selected parameters.

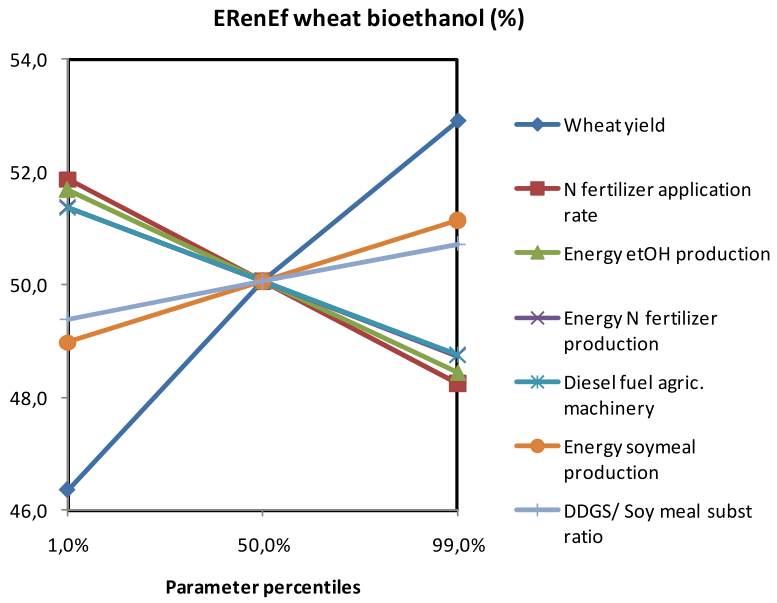


Fig. 4.9. Bioethanol from wheat (subst. method): sensitivity of ERenEf to selected parameters.

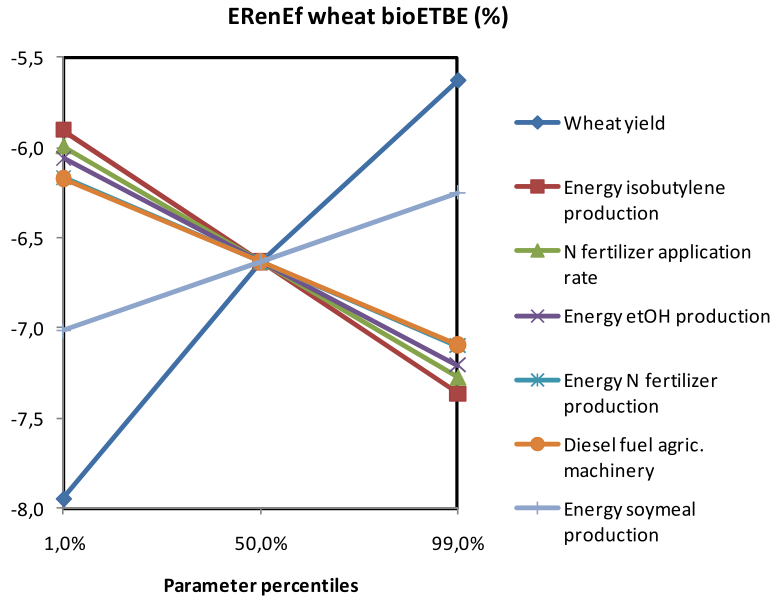


Fig. 4.10. BioETBE from wheat (subst. method): sensitivity of ERenEf to selected parameters.

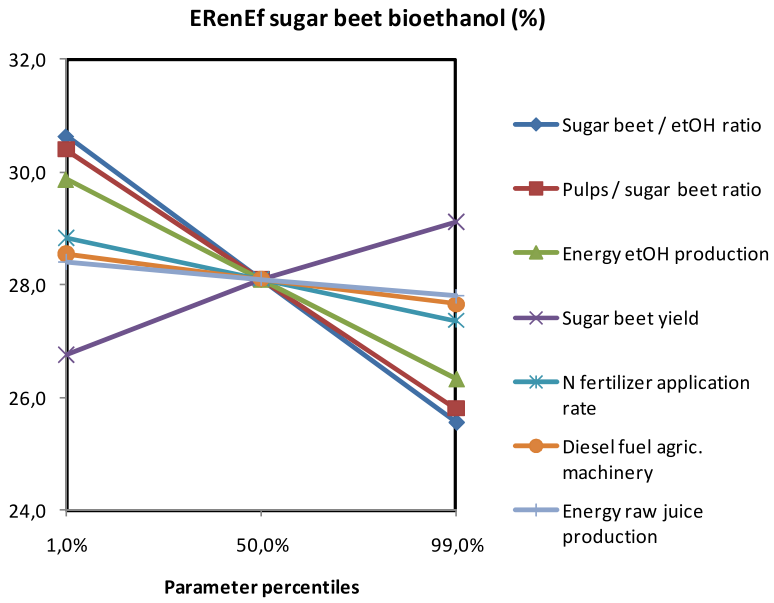


Fig. 4.11. Bioethanol from sugar beet (subst. method; bioethanol pathway): sensitivity of ERenEf to selected parameters.

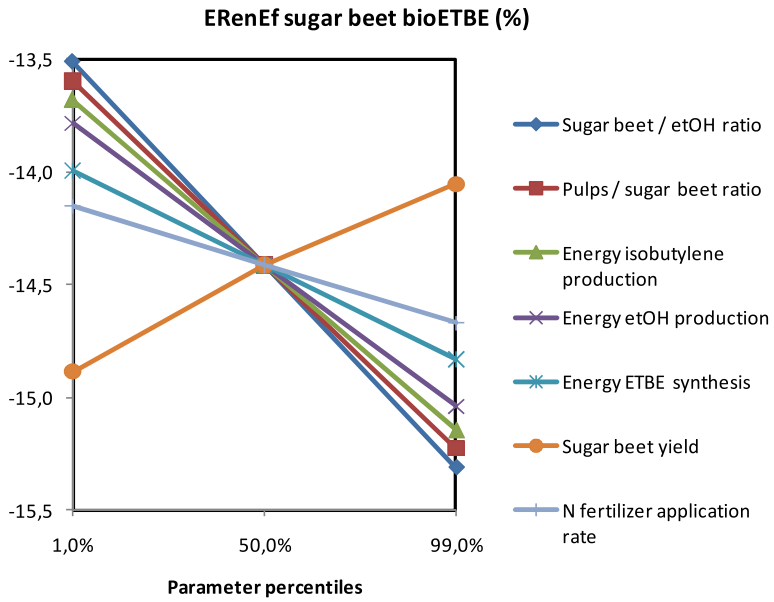


Fig. 4.12. BioETBE from sugar beet (subst. method; bioethanol pathway): sensitivity of ERenEf to selected parameters.

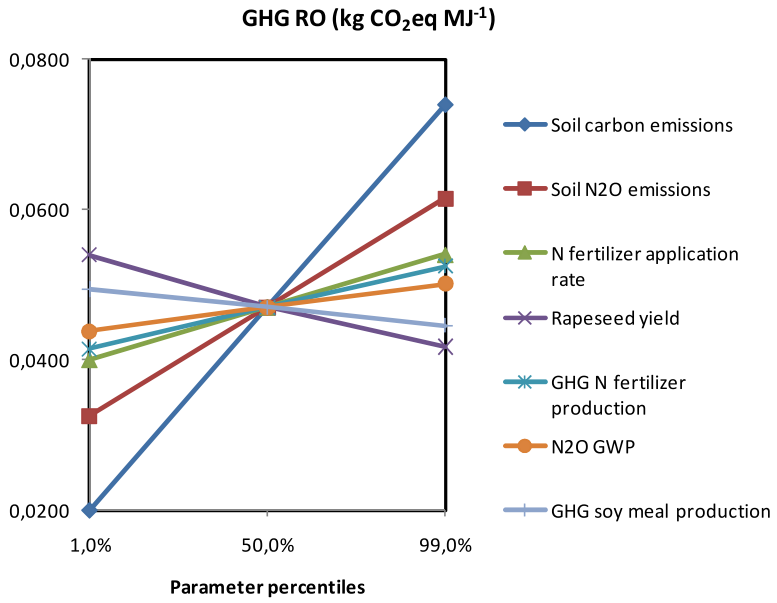


Fig. 4.13. RO (subst. method): sensitivity of GHG intensity to selected parameters.

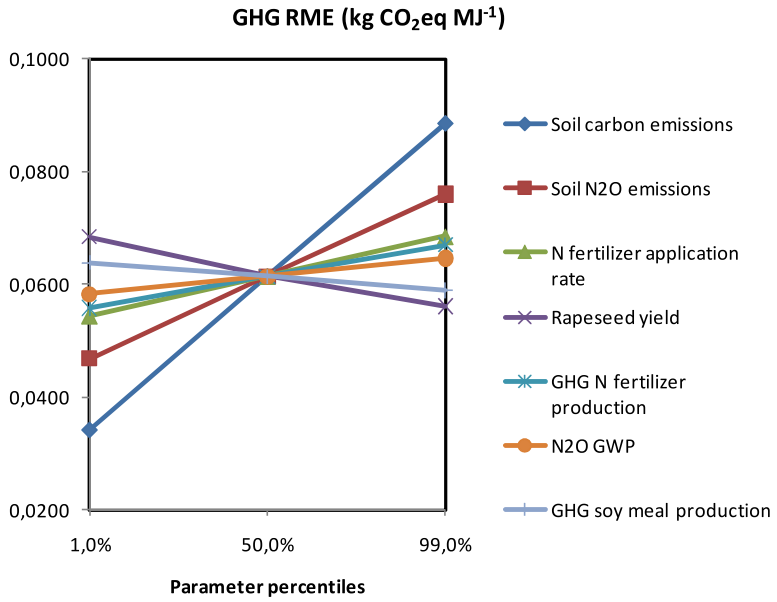


Fig. 4.14. RME (subst. method): sensitivity of GHG intensity to selected parameters.

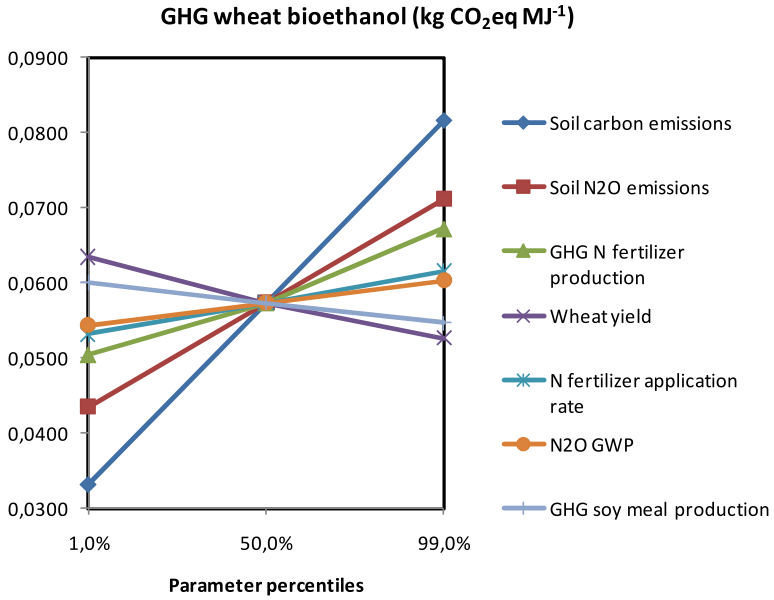


Fig. 4.15. Bioethanol from wheat (subst. method): sensitivity of GHG intensity to selected parameters.

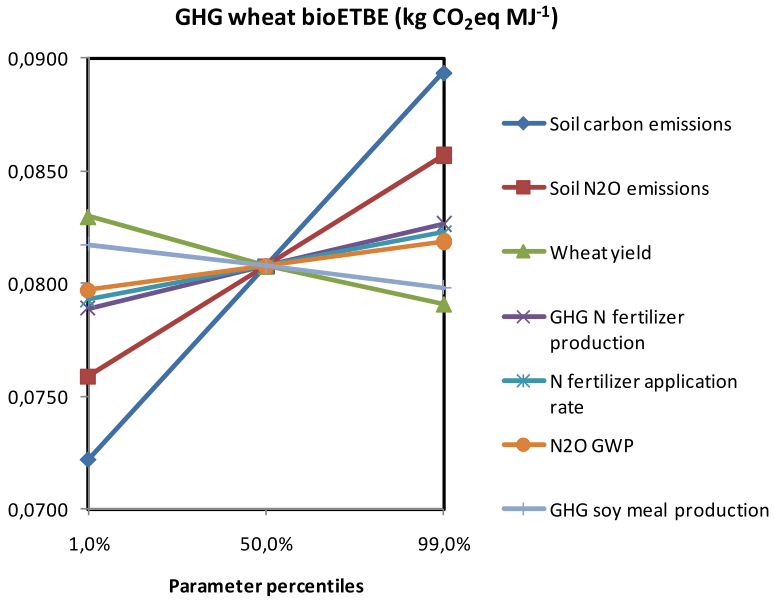


Fig. 4.16. BioETBE from wheat (subst. method): sensitivity of GHG intensity to selected parameters.

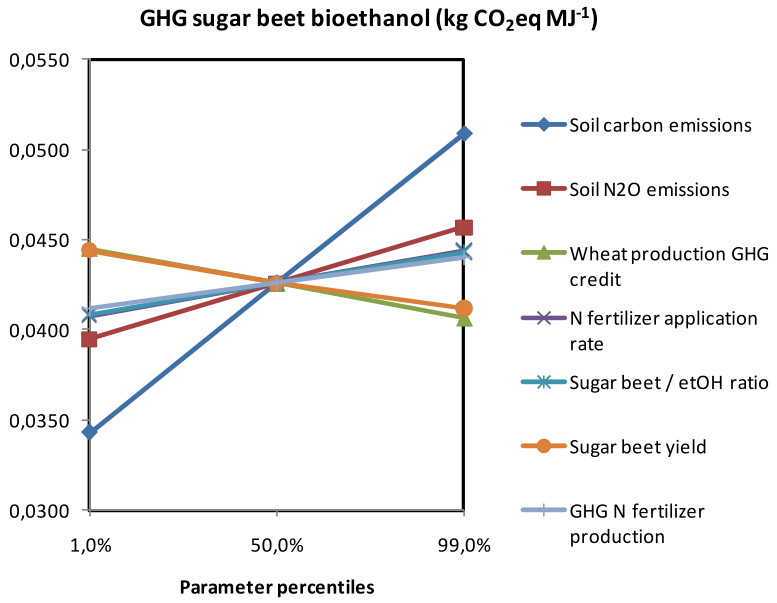


Fig. 4.17. Bioethanol from sugar beet (subst. method; bioethanol pathway): sensitivity of GHG intensity to selected parameters.

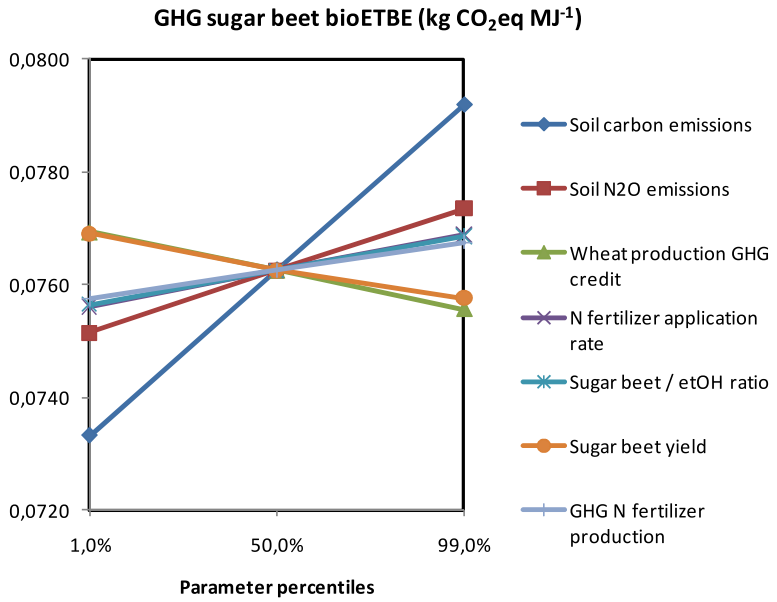


Fig. 4.18. BioETBE from sugar beet (subst. method; bioethanol pathway): sensitivity of GHG intensity to selected parameters.

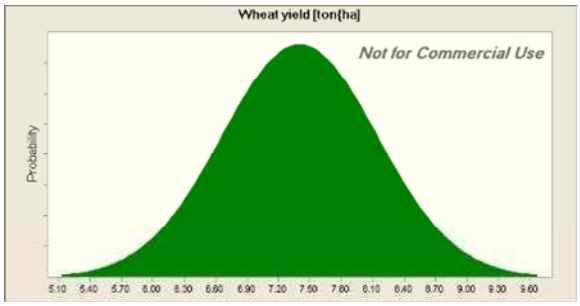
Single parameter sensitivity analysis has the merit of showing the influence of individual parameters (one at a time) on the overall biofuel system. The ability of investigating multiple uncertainties at once requires, however, a more powerful method, e.g. Monte-Carlo analysis (Johnson 2006; Plevin 2010). Monte-Carlo simulation is based on the repetition of hundreds to thousands individual model iterations – each using a randomly constructed set of parameter values – in order to assess the combined effect of the uncertainties of the most influential parameters on the model results (Schade and Wiesenthal 2011). Results based on Monte-Carlo uncertainty propagation are presented in chapter 6.

4.6. PRACTICAL ASPECTS OF MONTE-CARLO SIMULATION

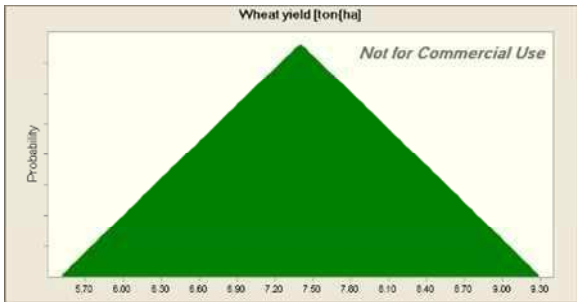
The following sections describe practical tests that have been conducted to assess the robustness of the developed biofuel life-cycle models. The models have been tested to different shapes of probability distributions and different sampling methods in Monte-Carlo simulation. The number of iterations required to achieve a specific level of precision in the results has also been calculated.

4.6.1. Shape of probability distributions

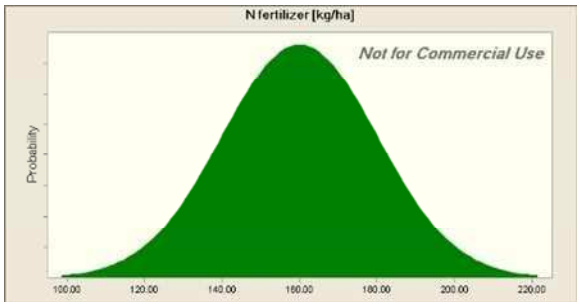
According to Plevin (2010), the specific shape of probability density functions (pdf) is not important qualitatively; instead, the choice of bounding values and the functional form of the model drive the results of a Monte-Carlo simulation. To understand the implications of selecting different probability density functions for input parameters, two alternative sets of pdfs (Fig. 4.19) have been assigned to important parameters affecting the energy renewability efficiency ERenEf and GHG intensity of wheat-based bioethanol (cf. Figs. 4.9 and 4.15): wheat agricultural yield and nitrogen fertilizer application rate (for ERenEf and GHG intensity); and carbon and N_2O emissions from soils (for GHG intensity). Alternative pdfs have the same 5^{th} and 95^{th} percentiles of the baseline pdfs. Tables 4.13 and 4.14 show that changing the pdf shape of parameters has little effect in the ERenEf and GHG intensity results, provided that bounding values (in this case, the 5^{th} and 95^{th} percentiles) are correctly chosen.



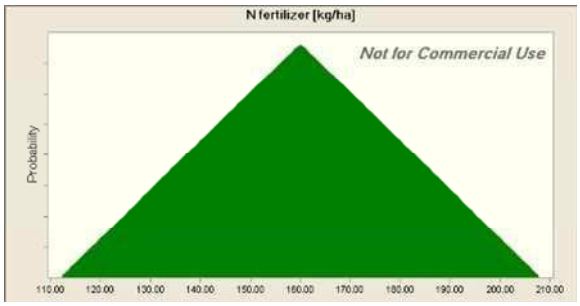
Normal (7.4; 0.74)



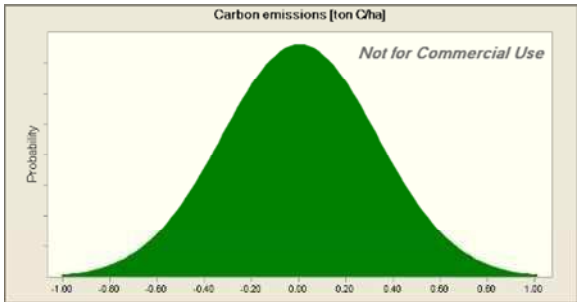
Triangular (5.6; 7.4; 9.2)



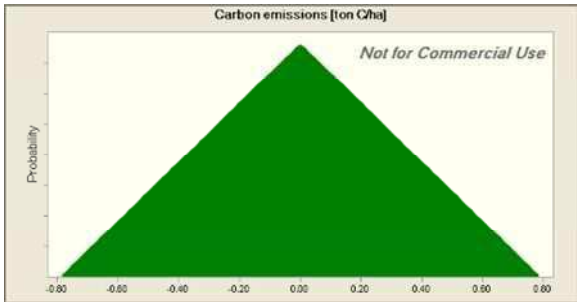
Normal (160; 20)



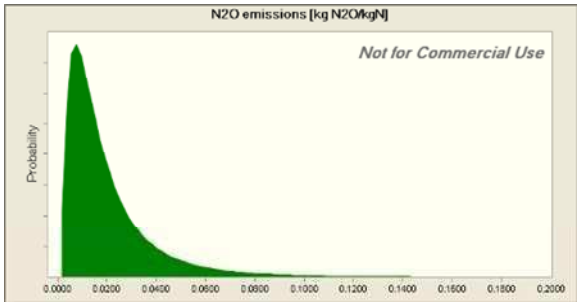
Triangular (111.9; 160.0; 208.1)



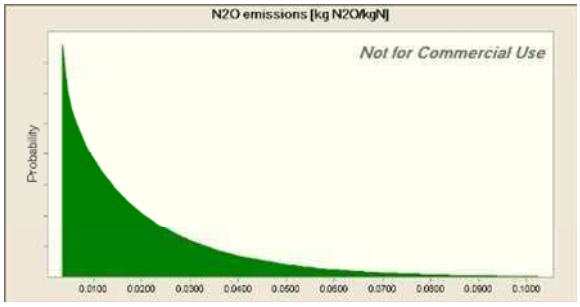
Normal (0.0; 0.33)



Triangular (-0.79; 0.00; 0.79)



Lognormal (0.0; 0.0208; 0.021)



Gamma (0.0; 0.02; 0.88)

(a) (b)

Fig. 4.19. Tested probability density functions (pdfs) for important parameters: (a) baseline pdfs [normal dist (μ; σ); lognormal dist (location; μ; σ)]; (b) alternative pdfs [triangular dist (min; likeliest; max); gamma dist (location; scale; shape)].

Table 4.13. ERenEf [%] of wheat-based bioethanol (no allocation) using different probability density functions for wheat yield and N fertilizer application rate.

Percentile	Baseline pdfs	Alternative pdfs
5^{th}	27.1	27.1
25^{th}	32.0	31.9
50^{th}	34.9	34.8
75^{th}	37.7	37.6
95^{th}	41.3	41.2

Table 4.14. Life-cycle GHG intensity of wheat-based bioethanol [g CO_2eq MJ^{-1}] (no allocation) using different probability density functions for important parameters (cf. Fig. 4.19).

Percentile	Baseline pdfs	Alternative pdfs
5^{th}	21.8	21.7
25^{th}	43.5	43.2
50^{th}	59.5	59.7
75^{th}	76.6	77.5
95^{th}	108.5	106.5

4.6.2. Number of iterations until convergence

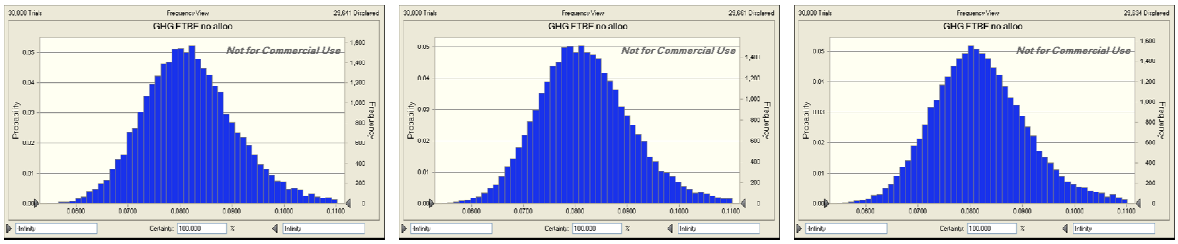
The number of iterations required to achieve convergence in the results has been assessed. The software used (Oracle Crystal Ball v.11.1) runs the Monte-Carlo simulation until specified precision limits are reached. At first, the average GHG intensity of wheat-based bioethanol has been used as the convergence criterion. In this case, GHG intensity values (for all treatment co-product approaches) in consecutive iterations must be within a specified interval. Four intervals were initially tested: ± 1; ± 0.5; ± 0.4; and ± 0.2 (g CO_2eq MJ^{-1}). When precision control limits were reached (for a confidence level of 95%) the software stopped the simulation and the number of iterations was recorded (Table 4.15). A precision around ± 0.4 or ± 0.5 g CO_2eq MJ^{-1} was considered appropriate for assessing the GHG intensity of biofuel systems and, according to data in Table 4.15, did not require an excessive number of iterations. Eventually, the value of 30000 iterations was chosen as the default value in Monte-Carlo simulation.

Table 4.15. No. of iterations required to reach a given level of precision in calculated biofuel GHG intensity.

Biofuel	Precision GHG [g CO_2eq MJ^{-1}] (95% confidence level)	No. of iterations until convergence
RO / RME	0.5	14050
	0.4	21350
Bioethanol (wheat)	1.0	1950
	0.5	6950
	0.4	11200
	0.2	43450
Bioethanol (sugar beet)	0.5	1100
bioethanol pathway	0.4	1500
Bioethanol (sugar beet)	0.5	33750
sugar pathway	0.4	52750

4.6.3. Sampling method

The software Crystal Ball provides two alternative methods for sampling variables during Monte-Carlo simulation: Monte-Carlo sampling (MCS), which uses random numbers to sample from a probability distribution; and Latin Hypercube sampling (LHS), which stratifies input probability distributions into strata of equal probability and then samples once from each stratum. Main disadvantage of LHS is the higher computational memory usage associated with stratification. On the other hand, the systematic procedure of LHS has the advantage of more precisely reproduce the shape of the sampled distribution, requiring a lower number of iterations than Monte-Carlo simulation. In other words, for the same number of iterations, LHS generates smoother output distributions compared to MCS. Fig. 4.20 shows three output probability distributions for the GHG intensity of wheat-based bioETBE in which the increased smoothness of the distribution is visible when LHS is used.



(a) (b) (c)

Fig. 4.20. Life-cycle GHG intensity of wheat-based bioETBE (no allocation) using (a) MCS; (b) LHS (100 intervals); and (c) LHS (500 intervals).

For the selected number of iterations, no differences have been calculated between Monte-Carlo and Latin Hypercube sampling methods, as shown in Table 4.16 for the bioETBE GHG intensity (no allocation). Due to the processing capabilities of modern computers and the "extreme speed" functionality of the Oracle Crystal Ball software which greatly increases simulation speed, the number of iterations is not a major issue. Therefore, Monte-Carlo sampling has been chosen as the default method in the simulations, with the advantage of lower memory requirements.

Table 4.16. Life-cycle GHG intensity of wheat-based bioETBE [g CO_2eq MJ^{-1}] (no allocation) using MCS and LHS methods.

Percentile	MCS	LHS (100 intervals)	LHS (500 intervals)
5^{th}	68.1	68.2	68.3
25^{th}	75.8	75.9	75.9
50^{th}	81.5	81.4	81.4
75^{th}	87.6	87.6	87.6
95^{th}	98.7	98.8	98.6

4.7. FOSSIL REFERENCE SYSTEMS

Table 4.17 gathers data on the energy requirement and GHG emissions of producing fossil diesel and gasoline. Data includes crude oil extraction, transport, fuel refinement

and distribution and excludes the energy delivered by the fuel itself and the GHG emissions of the combustion process. The uncertainty associated with the life-cycle energy and GHG emissions of fossil fuels is quantified using probability distributions as shown in Table 4.18. Data in table 4.18 have been used to compute the energy savings and avoided GHG emissions of biofuels over fossil fuels.

Table 4.17. Energy requirement and GHG emissions of producing fossil fuels.

Fossil fuel production (crude extraction, transport, and refining)				
	Min	*Med*	*Max*	
Fossil diesel				
Energy consumption		0.09		*ADEME (2002)*
[MJ MJ^{-1}]		0.12		*GM (2002)*
		0.11		*Mortimer et al. (2003)*
		0.16		*JEC (2004, 2007)*
	0.064	0.136	0.250	*Hekkert et al. (2005)*
GHG emissions		6.5		*ADEME (2002)*
[g CO_2eq MJ^{-1}]		10.2		*GM (2002)*
		8.6		*Mortimer et al. (2003)*
		14.2		*JEC (2004, 2007)*
	4	9	18	*Hekkert et al. (2005)*
Gasoline				
Energy consumption		0.15		*ADEME (2002)*
[MJ MJ^{-1}]		0.16		*GM (2002)*
		0.19		*Mortimer et al. (2003)*
		0.14		*JEC (2004, 2007)*
	0.111	0.22	0.37	*Hekkert et al. (2005)*
GHG emissions		10.5		*ADEME (2002)*
[g CO_2eq MJ^{-1}]		13.1		*GM (2002)*
		12.5		*JEC (2004, 2007)*
	8	15	26	*Hekkert et al. (2005)*

Table 4.18. Computed probability distributions for energy and GHG emissions of Fossil Diesel and Gasoline (includes the energy in the fuel and the emissions of combustion).

Parameter	**distribution**	**mean**	**std dev**
Fossil Diesel			
Energy Consumption [MJ MJ^{-1}]	Normal	1.14	0.04
GHG emissions (including combustion) [g CO_2eq MJ^{-1}]	Normal	82.0	3.0
Gasoline			
Energy Consumption [MJ MJ^{-1}]	Normal	1.19	0.05
GHG emissions (including combustion) [g CO_2eq MJ^{-1}]	Normal	84.0	3.0

4.8. CONCLUDING REMARKS

In this chapter, a life-cycle inventory for European biofuel chains has been conducted. Extensive data collection has been performed focusing on typical biofuel production systems in Europe and probability density functions have been assigned to parameters based on collected data. It is shown that biofuel production systems show significant sources of uncertainty concerning selection of parameter data and replacement options for co-products. Moreover, a sensitivity analysis highlighted the most influential parameters affecting the energy renewability and GHG intensity of the selected biofuels. One of the most important aspects – direct land use change – is thoroughly discussed in chapter 5.

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5. Modeling Direct Land Use Change

"Many prior studies have acknowledged but failed to count emissions from land-use change because they are difficult to quantify."

Searchinger et al. (2008)

"It is also essential that areas of uncertainty such as impacts on soil carbon stocks and fluxes are included in LCA assessments, and that further research is conducted to enable a robust calculation of impacts under different land-use change scenarios."

Whitaker et al. (2010)

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5. MODELING DIRECT LAND USE CHANGE

5.1. PURPOSE AND SCOPE

One of the most important issues affecting the GHG balance of biofuel systems, as demonstrated in the systematic review of chapter 3, is the inclusion of soil carbon emissions from direct land use change. A robust calculation of soil carbon fluxes when conducting biofuel life-cycle studies is required to assist in the development of soil management practices that protect existing soil carbon stocks and promote future sequestration (Ostle et al. 2009; Whitaker et al. 2010). A thorough discussion on soil carbon exchange associated with direct land use change follows.

Field observations and modeling studies show that land use and land use change significantly affect soil carbon stocks (Ostle et al. 2009). This issue can be addressed in several ways. For example, the IPCC (2006)[1] guidelines present methodologies for estimating GHG emissions from different land use categories, namely forest land, cropland, grassland, wetlands, settlements, and other land. The mathematical

[1] The 2006 IPCC Guidelines for National Greenhouse Gas Inventories provide internationally agreed methodologies intended for use by countries to estimate GHG inventories to report to the UNFCCC.

specification of the methods and data concerning emission factors is provided by the IPCC to generate the estimates. Three levels of detail are possible, from tier 1 (default method) to tier 3 (most detailed), depending on the availability of resources and the importance of the category of emissions under consideration.

Some authors develop their own models. Examples include the CESAR model, developed by Vleeshouwers and Verhagen (2002) which calculates carbon fluxes from agricultural soils in Europe taking into account crop, climate and soil types. Another example is the RothCUK dynamic modeling system, which estimates soil carbon fluxes from mineral soils in the UK caused by changes in climate, land use and land management (Fallon et al. 2006). Other authors evaluate the effect of land use change in soil carbon stocks based on long-term experiments, empirical models, soil carbon databases and data collated from literature surveys (see e.g. Boiffin et al. 1986; Bradley et al. 2005; Bernard and Prieur 2007).

In 2008, the European Commission published a proposal for a renewable energy directive with a methodology for calculating biofuel GHG emissions (EC 2008). In this proposal, soil carbon stock changes caused by land use change could be derived in a straightforward manner using a simplified table in which land use change data was highly aggregated. For example, set-aside land, non-permanent grassland and arable land were all assigned the same carbon stock (82 t C ha^{-1}).

More recently, in June 2010, specific guidelines for the calculation of land carbon stocks under the purpose of the renewable energy directive 2009/28/EC (EPC 2009) have been published (EC 2010a, 2010b, 2010c). These guidelines draw on the IPCC (2006) methodology[2] for national greenhouse gas inventories and provide detailed data for the calculation of soil carbon stocks under different land uses and cultivation practices. Nevertheless, EC (2010a) data is deterministic, not accounting for uncertainty ranges in parameters, as opposed to the 2006 IPCC guidelines from which they derive.

This chapter estimates soil carbon fluxes associated with land use change based on three different approaches. The main objective is to calculate soil carbon exchange (ΔC_{LUC})

[2] The guidelines draw on the IPCC (2006) tier 1 methodology, but include several simplifications in order to be readily applicable by economic operators.

values for specific land use change scenarios, including uncertainty data, and compare outcomes between addressed approaches. For each LUC scenario, uncertainty sources comprise soil management practices and geographic region (soil type and climate). With the aim of assessing the importance of soil carbon exchange in the overall GHG balance of biofuel systems, life-cycle results for two biofuel chains – rapeseed oil and wheat-based bioethanol – are also presented.

The chapter is divided in six sections, including this introduction. Section 5.2 describes the key issues that affect the process of soil carbon exchange. Section 5.3 presents an approach to direct LUC modeling based on the IPCC methodology and literature data collated prior to publication of directive 2009/28/EC. Section 5.4 models ΔC_{LUC} based on data from the European Commission decision 2010/335/EU (EC 2010a) and quantifies the overall uncertainty associated with each specific LUC scenario – e.g. from grassland to cropland – irrespective of the cultivation practices adopted. Section 5.5 presents a hybrid approach, in which ΔC_{LUC} for a specific set of selected land use change scenarios is calculated with deterministic data from EC (2010a) plus error ranges provided by IPCC (2006); this hybrid method is then compared with the results obtained if only deterministic data from EC (2010a) guidelines were used. Finally, section 5.6 concludes.

5.2. KEY ISSUES AFFECTING SOIL CARBON EXCHANGE

Important issues that affect the process of soil carbon exchange due to land use change are described in the following paragraphs.

LAND USE. Land use plays a central role in the life-cycle modeling of biofuels. Acreage expansion includes substituting previous crops with energy crops or converting uncultivated land – namely grasslands or land from set-aside programs – to biofuels (Fonseca et al. 2010). Land use conversion, however, promotes soil carbon exchange. On the other hand, higher crop yields on current arable land due to more intensive production methods may require increased inputs of capital, labor and materials, such as fertilizers (Melillo et al. 2009; Bergsma et al. 2007). This may lead to several environmental consequences, namely increased leaching of nutrients, nitrous oxide

emissions, pesticide contamination and loss of soil carbon (Kløverpris et al. 2008a; Fonseca et al. 2010). However, intensification reduces land use conversion and, if achieved without additional fertilizer inputs, a positive effect can be realized in terms of GHG emissions (Croezen et al. 2010). Elobeid et al. (2010), for example, state that yield growth is imperative for the long-term potential for first-generation biofuel expansion if land extensification is to be minimized. On the other hand, Keeney and Hertel (2010) point out the role of crop yield growth as a way of avoiding significant cropland conversion as the most controversial issue in the debate of agricultural land conversion versus GHG accounting of biofuels. In the IPCC (2006) guidelines and in the European Commission decision 2010/335/EU (EC 2010a) land use factors F_{LU} are used, which reflect the difference in soil organic carbon associated with the type of land use considered and the standard soil organic carbon SOC_{st}.

GEOGRAPHIC REGION. The geographic region is one of the key aspects for assessing GHG emissions at the cultivation stage, since climate and soil type are two important factors affecting the calculation of land carbon stocks. Main biofuel producers in Europe are Germany and France (EurObserv'ER 2010). Other European regions with diverse climate regions that cultivate energy crops include Poland, the United Kingdom, Czech Republic and Denmark (FAO 2010). According to the classification presented in EC (2010a)[3], the climate regions that characterize these countries are: (i) cool temperate, moist, CTM (Germany, Poland, UK and Czech Rep.); (ii) warm temperate, moist, WTM (France); (iii) cool temperate, dry, CTD (Germany, Poland, UK and Denmark, Schmidt 2007); and (iv) warm temperate, dry, WTD (France). Concerning soil types, EC (2010a) shows that high activity clay soil is the most representative soil type for countries involved in the cultivation of energy crops. Active soils are also indicated as the most likely soil type to be converted to arable cropping by JEC (2007). Table 5.1 shows the standard carbon stock values for the selected soil types and climate regions, according to EC (2010a).

[3] Climate region data layers and soil type data layers based on GIS mapping are available through the European Transparency Platform established by Directive 2009/28/EC. Detailed data can be found in: http://eusoils.jrc.ec.europa.eu/projects/RenewableEnergy/ (accessed December 2010).

Table 5.1. Standard soil carbon stock in the 0-30 cm topsoil layer (t C ha^{-1}) [EC 2010a].

Climate	High activity clay soils (HACS)	Sandy soils (SS)
Cool temperate, moist (CTM)	95	71
Warm temperate, moist (WTM)	88	34
Cool temperate, dry (CTD)	50	34
Warm temperate, dry (WTD)	38	19

SOIL MANAGEMENT PRACTICES. Alternatives in soil management practices are quantified through a factor (F_{MG}) that reflects the difference in soil organic carbon associated with the main management practice in comparison to the standard soil organic carbon SOC_{st} (IPCC 2006; EC 2010a). A first broad distinction is made between cropland and grassland. Concerning cropland, IPCC (2006) and EC (2010a) differentiate three alternative management practices –full-tillage; reduced or low-tillage; and no-till–, based on the level of soil disturbance during cultivation, respectively substantial, reduced or minimal. As regards grassland, IPCC (2006) and EC (2010a) consider four different management scenarios: (i) improved, which represents sustainably managed grassland with moderate grazing pressure and receiving at least one improvement; (ii) nominally managed, which is equivalent to improved but without significant management improvements; (iii) moderately degraded, which represents overgrazed or moderately degraded grassland; and (iv) severely degraded, which implies major long-term loss of productivity and vegetation cover.

CARBON INPUT TO SOIL. The level of carbon input to soil may differ depending on the return of crop residues to the field and the adoption of other practices such as addition of animal manure or use of perennial grasses in crop rotations (EC 2010a). A classification in four categories is presented in IPCC (2006) and EC (2010a), which takes also into account the level of mineral fertilization and the use of nitrogen-fixing crops in rotation: low; medium; high; and high with manure carbon input cropping systems. An input factor F_I 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} (IPCC 2006; EC 2010a).

Residues not incorporated in the field may be used as heating fuel and to generate electricity (Richards 2000; Mortimer et al. 2003; Powlson et al. 2008), as animal fodder (Dewulf et al. 2005), for cattle, horses and sheep bedding (Scarlat et al. 2010), for mulching of horticultural crops and mushroom production (Powlson et al. 2008; Scarlat et al. 2010). Straw burning is not permitted in most countries due to environmental regulations (Powlson et al. 2008), and the industrial uses of straw – pulp and paper production or as insulating material for buildings – are estimated to account for a very small fraction (around 1%) of total production (Scarlat et al. 2010).

5.3. dLUC MODELING BASED ON LITERATURE DATA[4]

This section presents a comprehensive assessment of different land use change scenarios – from former (i) agricultural land, (ii) set-aside land, and (iii) grassland, to actual use for cropland – based on data collated prior to publication of EC (2010a) guidelines on the calculation of land carbon stocks. The literature survey that has been conducted and the ranges selected for carbon stock change data aggregation are presented in the following paragraphs. An application to the life-cycle of rapeseed oil, which includes the assessment of rapeseed cultivation according to these LUC scenarios, closes the section.

5.3.1. Land use change scenarios

CROPLAND TO CROPLAND. Some approaches assume that switching between different crops is neutral in terms of soil carbon stock change. For example, in the methodology for calculating GHG emissions from production and use of biofuels presented in EC (2008), carbon stocks for cropland amount to an equilibrium value of 82 t C ha^{-1}, irrespective of the crop considered. Likewise, Piñeiro et al. (2009) have estimated no net gain or loss of soil carbon in corn production starting from agricultural land and Jungbluth et al. (2007) have not considered any soil carbon change from land use conversion when switching from cropland to rapeseed cultivation under German conditions. Data on carbon sequestration and emission for several crops, including rapeseed, is also presented in other studies, namely Boiffin et al. (1986), Wylleman

[4] Part of this section has been published in Malça & Freire (2010a) and Malça & Freire (2009c).

(1999), Arrouays et al. (2002) and Bernard and Prieur (2007). From the available data, a comparison of soil carbon fluxes between different crops can be made. Using the methodology proposed in Arrouays et al. (2002), carbon emissions from soil can be calculated respectively as 0.08 ± 30% and 0.16 ± 30% t C ha^{-1} yr^{-1} when rapeseed substitutes barley and wheat in a crop rotation scheme.

SET-ASIDE LAND TO CROPLAND. According to the methodology proposed in EC (2008), changing from set-aside to cropland has no net effect on the soil carbon content. The IPCC (2006) guidelines for national GHG inventories also provide generic methodologies for estimating carbon stock changes associated with different land use conversions and, in particular, for the conversion from a temporary set-aside of annually cropland to cultivated land. Assuming a reference carbon stock of 50 t C ha^{-1} – for soils with high activity clay minerals, the most likely soil type to be converted to arable cropping, according to JEC (2007) – and a nominal error of ± 90% as indicated by the IPCC, an annual lost in soil carbon stock of 0.325 ± 90% t C ha^{-1} yr^{-1} is calculated for the conversion of a temporary set-aside land to a full tillage cultivated land in a temperate, dry climate. Piñeiro et al. (2009) have estimated changes in soil organic carbon from lands maintained in set-aside and later converted for corn ethanol production. Using a 20-year allocation time period for LUC emissions, an annual loss of 0.35 t C ha^{-1} yr^{-1} is calculated.

GRASSLAND TO CROPLAND. Increasing the arable area at the expense of grassland releases soil organic carbon to the atmosphere (Ogle et al. 2005). Although this only happens once, when native grasslands are cropped for the first time, the effect is very large and long-lasting (JEC 2007). For example, the use of the IPCC (2006) methodology shows that switching from nominally managed (i.e. non-degraded or improved) grassland to cropland results in an annual loss in the organic carbon stock in soils of 0.5 t C ha^{-1} yr^{-1}. Assuming that previous land use was improved grassland, the carbon loss would increase to 0.85 t C ha^{-1} yr^{-1}. Piñeiro et al. (2009) have also evaluated the conversion of native grasslands to corn cultivation: emissions of approximately 60 t CO_2eq ha^{-1} have been computed in a 20-year period, i.e. an average soil carbon loss of 0.8 t C ha^{-1} yr^{-1} between previous and new soil carbon stabilization levels. Guo and

Gifford (2002) have reviewed the effects of various land use changes on soil carbon stocks. Using a meta-analytical approach, these authors have concluded that after conversion from pasture to crop, soil C stocks decline between ca. 45% and 60% in the first 30 years after conversion. Assuming a reference soil carbon stock of 50 t C ha^{-1}, as explained before (JEC 2007; IPCC 2006), carbon losses of 0.75 to 1.0 t C ha^{-1} yr^{-1} can be calculated. Soussana et al. (2004) have assessed soil organic carbon fluxes resulting from land use change, in particular between cropland and grassland, and including uncertainty: these authors have calculated a soil carbon stock reduction of 0.95 ± 0.3 t C ha^{-1} yr^{-1}. JEC (2007) has also evaluated the reduction in the carbon stored in the soil due to plowing up grassland: 73 t of CO_2 are emitted per hectare in a 20-year period, with an uncertainty range of more than 50%, i.e. 1.0 ± 50% t C ha^{-1} yr^{-1}.

Literature data on the carbon stock changes associated with the three alternative land use change scenarios and the corresponding ranges that have been considered are listed in Table 5.2.

Table 5.2. Changes in soil carbon stocks (ΔC_{LUC}) for three land use change scenarios: literature data and ranges considered.

LUC scenario	ΔC_{LUC} [t C ha^{-1} yr^{-1}]	Sources	Ranges used [t C ha^{-1} yr^{-1}] (1)	Ranges used [g CO_2eq MJ^{-1}] (2)
Cropland → Cropland			**0-0.18**	**0-12.1**
	0	Jungbluth et al. (2007); Piñeiro et al. (2009); EC (2008)		
	0.08-0.16 ± 30%	Arrouays et al. (2002); Boiffin (1986); Wylleman (1999); Bernard & Prieur (2007)		
Set-aside → Cropland			**0.1-0.5**	**6.7-33.5**
	0	EC (2008)		
	0.24 ± 30%	Bernard & Prieur (2007)		
	0.325 ± 90%	IPCC (2006)		
	≈ 0.35	Piñeiro et al. (2009)		
Grassland → Cropland			**0.5-1.25**	**33.5-83.8**
	0.50-0.85 ± 90%	IPCC (2006)		
	≈ 0.80	Piñeiro et al. (2009)		
	0.75 – 1.0	Guo and Gifford (2002)		
	0.95 ± 0.3	Soussana et al. (2004)		
	1.0 ± 50%	JEC (2007)		

(1) Uniform probability distributions have been used for soil carbon stock changes; (2) Calculated using an average rapeseed productivity of 3.5 t $ha^{-1}yr^{-1}$, an oil extraction rate of 42%, and LHV_{RO}=37.2 MJ kg^{-1}.

5.3.2. GHG intensity results

Life-cycle GHG emissions of rapeseed oil for the three LUC scenarios addressed are shown in Fig. 5.1. 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. Weight, energy and economic parameters, as well as GHG emission substitution credits, were taken from Malça and Freire (2009a, 2009c). Parameter uncertainty is very high for all LUC scenarios. In particular, in the "cropland to cropland" scenario, parameter uncertainty clearly overcomes the differences between calculated mean values for the several approaches used for dealing with co-products. Conversion of grassland to cropland induces the highest ΔC_{LUC} emissions and thus the highest life-cycle GHG emissions for rapeseed oil. With the substitution method, RO GHG emissions are higher than (displaced) fossil diesel emissions (82 g CO_2eq MJ^{-1} on average).

As discussed in chapter 3, assumptions about previous and actual land uses in a life-cycle study are a relevant issue and shall be clearly reported. Preferably, the GHG contribution of LUC shall be presented in a transparent and disaggregated way from the rest of the life-cycle (Menichetti and Otto 2009). The importance of soil carbon exchange associated to LUC in the life-cycle GHG intensity of rapeseed oil can be assessed by comparing the ranges used for ΔC_{LUC} (in g CO_2eq MJ^{-1}, right column of Table 5.2) with the results shown in Fig. 5.1. On average, ΔC_{LUC} represents 12.4%, 31.4% and 56.3% of the GHG intensity of RO (non-allocated results).

Table 5.3 lists which parameters are most significant in the overall uncertainty of RO GHG emissions for the three LUC scenarios under study. A gradually higher contribution of soil carbon to the variance of greenhouse gas emissions is clearly shown, as we move from the "cropland to cropland" to the "set-aside to cropland" scenario and finally to the "grassland to cropland" scenario. Nitrous oxide emissions from cultivated soil also contribute significantly to the variance of GHG emissions.

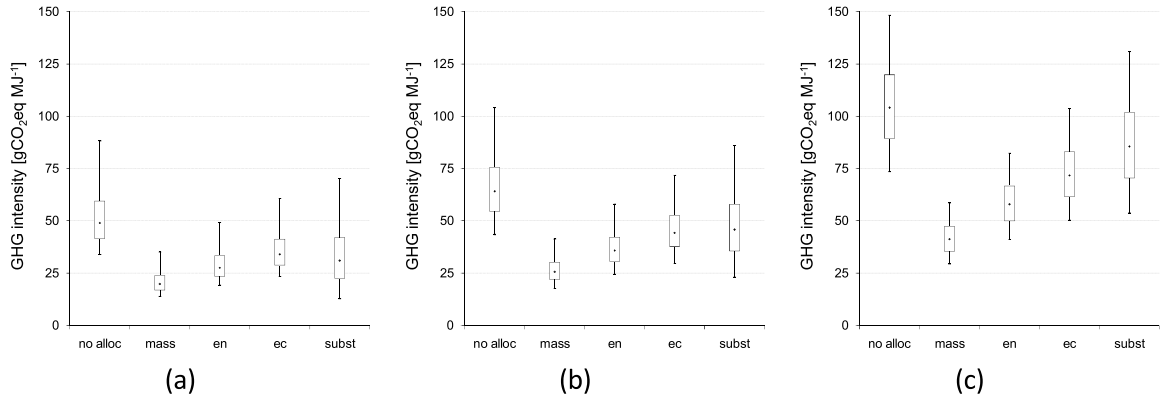


Figure 5.1. Life-cycle GHG intensity of rapeseed oil with the following LUC scenarios: (a) "cropland to cropland"; (b) "set-aside to cropland"; (c) "grassland to cropland". The boxes show the interquartile range, the mark is the median and the ends of the whiskers are the 5th and 95th percentiles. Same notation is used in the remaining figures.

Table 5.3. Contribution of input data to the variance of RO GHG emissions (in %).

Scenario/Parameter [(1)]	No allocation	Mass	Energy	Economic	Substitution
Cropland → Cropland					
Soil N_2O emissions	63.6	63.6	63.6	61.2	54.6
N_2O GWP	13.1	13.2	13.2	12.7	10.8
Soil carbon emissions (dLUC)	7.7	7.7	7.7	7.1	6.3
Soy meal subst. credit	n/a	n/a	n/a	n/a	15.2
Set-aside → Cropland					
Soil N_2O emissions	47.7	47.7	47.7	46.4	43.0
Soil carbon emissions (dLUC)	28.8	28.8	28.8	27.1	25.1
N_2O GWP	9.5	9.6	9.6	9.4	8.6
Soy meal subst. credit	n/a	n/a	n/a	n/a	10.6
Grassland → Cropland					
Soil carbon emissions (dLUC)	52.2	52.2	52.2	48.3	49.0
Soil N_2O emissions	29.2	29.2	29.2	27.0	27.9
N_2O GWP	6.4	6.4	6.4	6.1	6.0
Soy meal subst. credit	n/a	n/a	n/a	n/a	5.6

[(1)] Cut-off criterion of 5%; n/a: not applicable.

5.4. dLUC MODELING CAPTURING VARIABILITY OF AGRICULTURAL PRACTICES

In this section, direct land use change scenarios are addressed taking into account the variability associated to different agricultural practices. Within each LUC scenario, potential agricultural practices in terms of soil management and carbon inputs to soil are considered. The approach uses data published in EC (2010a) to quantify F_{LU} (land use),

F_{MG} (soil management), and F_I (levels of carbon input to soil) factors. The calculation method uses the equations presented in section 2.3.3. This model works as a generic model for rapeseed cultivation in Europe, embracing all possible agricultural practices for each LUC scenario. It gives, therefore, a wide perspective of the uncertainty that can be found in rapeseed cultivation in Europe.

5.4.1. Land use change scenarios

BACKGROUND. For the purposes of this section, two main reference land uses are considered – cultivated cropland and grassland – with appropriate F_{LU} values taken from EC (2010a): from 0.69 to 0.80 for cultivated cropland (variation depends on the climate region considered); and 1.0 for grassland.

Concerning rapeseed cultivation in Europe, several climate regions can be found (EC 2010a): CTM, WTM, CTD, and WTD (cf. Table 5.1). Calculation of soil carbon exchange ΔC_{LUC} is conducted for these four climate regions. High activity clay soil is the most representative soil type for rapeseed cultivation in Europe. Nevertheless, rapeseed cultivation in sandy soils is also reported in the literature. For example, Bonari et al. (1995) conducted a 3-year rapeseed cultivation test on a very sandy soil in Italy. The very low water retention capacity of this type of soils favors crops with an autumn-spring growth cycle. Winter rapeseed in particular showed a very good adaptability to the test environment. Rathke et al. (2005) and Rathke and Diepenbrock (2006) also conducted a 6-year field experiment with winter rapeseed on fertile sandy loess in central Germany to evaluate the effect of different fertilization types and application rates as well as different preceding crops. Therefore, two distinct scenarios in terms of soil type for rapeseed cultivation are addressed in this section: high activity clay soils (HACS) and sandy soils (SS).

Concerning soil management practices in cropland, the three alternatives of EC (2010a) are taken into account when modeling the reference land use. As regards the actual land use –rapeseed cultivation–, the crop can be managed using any of those methods. Bonari et al. (1995), for example, compared the effects of conventional plowing and

minimum tillage on winter rapeseed in a sandy soil in center Italy. Despite the remarkable reductions in working time, fuel consumption and energy requirement associated with minimum tillage, rapeseed yields under conventional and minimum tillage never differed significantly. Similarly, Hocking et al. (2003) assessed the influence of different tillage treatments – conventional, one-pass and no-till – on rapeseed cultivars for two seasons at high and low rainfall sites in Australia and concluded that tillage had little effect on seed yields. Nevertheless, conservation-farming systems involving no-till or one-pass tillage reduced the risk of soil degradation and saved time and land-preparation costs. Other authors also concluded that the adoption of no tillage and/or minimum tillage results in crop yields that do not differ significantly from those obtained using conventional plowing (Vez and Vullioud 1971; Christian and Bacon 1990). Depending on the climate region and type of management practice, F_{MG} values for cropland in this analysis may vary between 1.0 and 1.15 (EC 2010a). Concerning grassland, the four management classes indicated in EC (2010a) – improved, nominally managed, moderately degraded, and severely degraded – are considered, with F_{MG} values ranging from 0.7 to 1.14 (EC 2010a).

Next paragraphs detail the aspects affecting the level of carbon input to soil. Concerning the production level of different agricultural crop residues in the EU-27, rapeseed is the 4th most important crop (Scarlat et al. 2010). Main applications for crop residues (other than for energy production or burned in the field) include incorporation into soil, animal feed and bedding, surface mulching in horticulture, mushroom cultivation and industrial uses (Powlson 2007). Several studies point out the incorporation of straw and other crop residues in the soil as a farm management activity. Rape straw, for example, is mostly ploughed back into the soil, because it contains nitrogen and minerals taken up by the crop and is needed to improve the organic content of the soil (SenterNovem 2005a; JEC 2007; UFOP 2008; Börjesson and Tufvesson 2011). Other authors, however, indicate removal rates for rapeseed straw ranging between 30% (Nikolaou et al. 2003) and 50% (Newman 2003). Residue removal rates depend on several factors, namely equipment limitations, crop yields and environmental requirements. In order to be sustainable, the collection of crop residues must protect soil from erosion while maintaining the soil organic carbon content (Scarlat et al. 2010). Besides differences in residue removal

rates, the residue-to-seed ratio is in itself subject to variability, with typical values for rapeseed in the range of 1.4 to 2.0 kg kg^{-1} (Scarlat et al. 2010).

Another aspect concerning soil inputs is the possibility that manure substitutes for mineral fertilization, as shown e.g. by Rathke et al. (2005) and Rathke and Diepenbrock (2006). These authors evaluated the energy efficiency, seed yield, seed oil and protein content of winter rapeseed to varying application rates of calcium ammonium nitrate and cattle manure slurry. Main conclusion is that the N fertilization rate had the strongest influence on the results, whereas the type of fertilizer and the preceding crop only had a small effect. The energy efficiency, seed yield and protein content of winter rapeseed responded well to different N management strategies, with high energy outputs and seed yield and protein content for high rates of N applied. On the other hand, increasing N rates caused a decrease in oil content. The energy output per hectare associated with rapeseed was lower when slurry was used instead of mineral fertilizer, but the difference could be neglected for an application rate of 160 kg N ha^{-1}. Concerning the other parameters, mineral N fertilization versus slurry application resulted in higher rapeseed yield and crude protein content, but lower oil content. Similarly, Gao et al. (2010) investigated the effects of different nutrient sources (urea fertilizer and manure) and different levels of N application on rapeseed yield and oil content. These authors concluded that nutrient applications were not necessary to increase rapeseed yield in fertile fields. Furthermore, at similar N levels, total oil content in rapeseed with fertilizer use was sometimes lower than that with manure application. Kazemeini et al. (2010) also evaluated the implications of N fertilization and the use of manure on rapeseed yield during two growing seasons (2006/07 and 2007/08) in Iran. Results showed a beneficial effect of manure on the reduction of N fertilizer rates, with maximum rapeseed yields achieved when 50% of the required N fertilizer is replaced by manure application.

As regards other cultivation practices, Bona et al (1999) evaluated the possibility of applying low-input management to oilseed crops, including rapeseed, in order to reduce the environmental impact associated with the high levels of agronomic inputs that characterize intensive management of soils. These authors showed that increasing plant

density, i.e. increasing the number and length of plant roots in the soil, is a key factor in maximizing nitrogen uptake. This cropping technique is therefore a strategy to reduce nitrogen fertilization, with consequent energy savings and reduced probability of nitrate leaching. A different technique was used by Hocking et al. (2003), who tested the response of rapeseed to different fertilizer placements at sowing: placed with the seed; broadcast; and banded to the side and 3 cm below the seed. The main concern underlying this test was to avoid chemical injury to germinating seeds. These authors concluded that, for the same level of fertilization, separation of seeds from fertilizer has the potential for producing high yielding rapeseed crops. Input factors F_I may vary from 0.92 to 1.44 and from 1.0 to 1.11, for cropland and grassland, respectively (EC 2010a).

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 (2010a) concerning the selection of the appropriate coefficients F_{LU}, F_{MG} and F_I for land use, management and carbon input practices for two land use categories – cropland and grassland – is followed.

LUC SCENARIOS. Six alternative scenarios of land use conversion from cropland or grassland to rapeseed cultivation are considered. Table 5.4 lists the reference and actual land uses in the scenario analysis. Scenarios are based on rapeseed cultivation in soils that were previously allocated to cropland or grassland. According to EC (2010c), land use change refers to changes in terms of land cover between the six land categories used by the IPCC. This dissertation follows a wider definition for LUC, which also includes changes of agricultural management activities. Fallow land – land set at rest for one or several years before being cultivated again – is included in the cropland category. Actual land use has been divided in two main categories: (i) cropland in which rapeseed may be cultivated with any of the main management practices and levels of carbon input indicated in EC (2010a), with the exception of manure application ("cropland 3" for labeling purposes in this section[5]); and (ii) cropland with any of the main soil management practices of EC (2010a) but the regular addition of animal manure

[5] The reason for this label has to do with cropland having three (3) different levels of carbon input to soil in this scenario. Similarly, reference land use "cropland 4" includes the four (4) different levels of carbon input considered in EC (2010).

("cropland 1"). The additional practice of using manure partially displaces mineral fertilizer use, as demonstrated by Kazemeini et al. (2010). Hence, the nitrogen fertilizer application rate of the "cropland 1" scenario was halved in relation to the "cropland 3" rate. Concerning reference (previous) land use, three distinct scenarios were developed: (i) "cropland 4" which covers all cropland management practices and carbon input levels specified in EC (2010a); (ii) "grassland 3" which includes improved, nominally managed and moderately degraded grassland[6]; and (iii) "grassland 1" which represents severely degraded grassland. In particular, a bonus of 29 g CO_2eq per MJ of biofuel can be attributed in the GHG calculation if evidence is provided that biomass cultivation on degraded land contributes to increase the soil carbon stock (EPC 2009). This bonus has been credited in the calculation. Several climate regions and soil types were also addressed, as shown in section 5.2.

Table 5.4. Reference and actual land uses considered in scenario analysis (in brackets: labels used in section 5.4).

REFERENCE LAND USE		ACTUAL LAND USE
Cropland: all management practices; all input variants *(cropland 4)*	→	Cropland: all management practices; all inputs except high with manure *(cropland 3)*
Grassland: improved, nominally managed; moderately degraded *(grassland 3)*	→	*(cropland 3)*
Grassland: severely degraded *(grassland 1)*	→	*(cropland 3)*
(cropland 4)	→	Cropland: all management practices; high with manure input *(cropland 1)*
(grassland 3)	→	*(cropland 1)*
(grassland 1)	→	*(cropland 1)*

[6] The three management levels included in this scenario motivate the label "grassland 3".

5.4.2. Probability distributions for soil carbon exchange

Taking into account six different land use change scenarios, four climate regions and two types of soil, as discussed in the previous sections, a total of 48 different alternative scenarios for rapeseed cultivation were assessed. The variability associated with different soil management and input practices within each scenario has been quantified through Monte Carlo simulation. The variations in land carbon stocks between reference (CS_R) and actual (CS_A) land uses were calculated using the coefficients proposed in EC (2010a) [equations 2.8 to 2.10] and appropriate probability density functions were assigned using a goodness-of-fit method in Oracle Crystal Ball software. For small sample sizes (less than 15 elements), histograms were created to assist in the selection of the appropriate probability distribution.

Fig. 5.2 and Table 5.5 show the probability distribution that best fits the variation of soil carbon stock (ΔC_{LUC}) in each scenario. Depending on the reference and actual land uses considered, the soil carbon content may increase or decrease with LUC (Hoefnagels et al. 2010). Positive values mean net carbon emissions from soil due to land use change, whereas negative values indicate carbon sequestration. Fig. 5.2 shows that:

- for equivalent scenarios, reference land use “cropland 1” has lower carbon emissions (or higher sequestration levels) than “cropland 3”, which is due to (i) the higher contribution of manure application to the carbon content of soils; and (ii) the lower mineral fertilizer application rates of “cropland 1”;
- a land use change from “grassland 3” to “cropland 3” (Fig. 5.2b) contributes to the highest carbon emissions, meaning that in general a change from grassland to cropland is the worst scenario in terms of GHG performance;
- when the reference land use is “grassland 1”, the carbon sequestration levels achieved with LUC are equivalent to those of the “cropland 4” scenario. This can be explained by the relatively low carbon content of the “grassland 1” scenario, which is severely degraded grassland;
- in general, uncertainty ranges are lower in Fig. 5.2(c) because the “grassland 1” reference land use is deterministic;

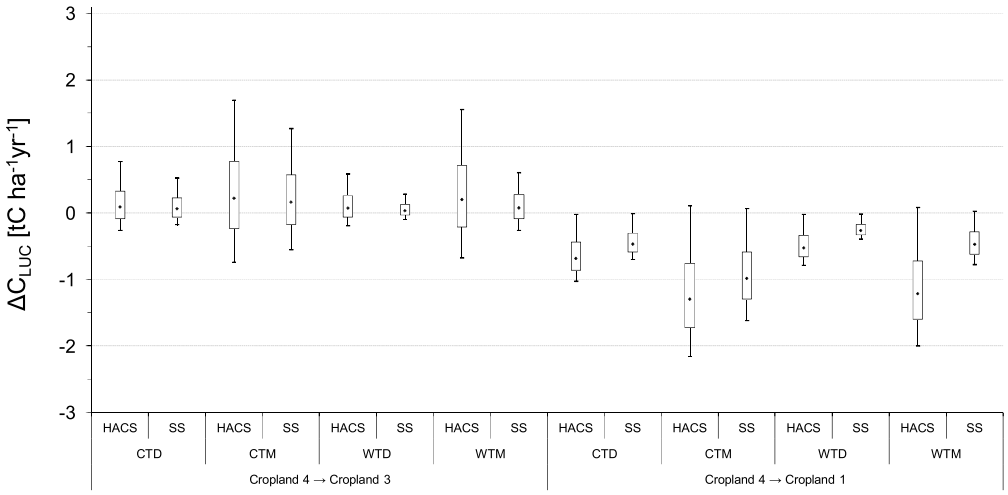
- uncertainty ranges are highest for the following combinations of climate region/ type of soil: (CTM,HACS); (WTM,HACS); and (CTM,SS). This is due to the highest standard soil carbon contents in these particular combinations (cf. Table 5.1).

Comparing the current approach with the approach used in section 5.3, it is shown that the former is more robust in evaluating soil carbon fluxes due to land use change. First, it distinguishes among soil types and climate regions. Secondly, it takes into account more land use categories. Thirdly, the assessment of uncertainty associated with different soil management and input practices for each (reference and actual) land use relies on more detailed data.

A comparison between soil carbon exchange values in Fig. 5.2 and ΔC_{LUC} ranges used in the approach of section 5.3 (based on literature data, cf. Table 5.2), shows that in general the "cropland 4 → cropland 3" and "grassland 3 → cropland 3" LUC scenarios (Fig. 5.2) agree, respectively, with the "cropland → cropland" and "grassland → cropland" LUC scenarios (section 5.3). One of the differences is the large uncertainty range in certain scenarios of Fig. 5.2 (cf. coefficient of variation CV), which is due to the higher uncertainty associated with different agricultural management practices, an aspect that has not been explicitly modeled in the approach of section 5.3. An additional difference is the higher median values of ΔC_{LUC} for the abovementioned combinations of climate and soil types –(CTM,HACS), (CTM,SS) and (WTM, HACS)–, which is explained by the high soil carbon stock of these scenarios.

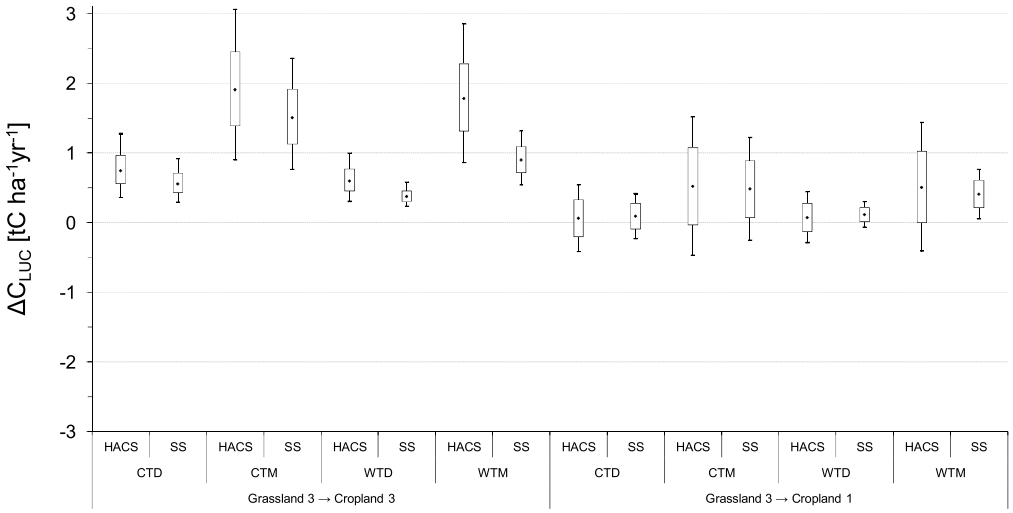
The soil carbon changes due to LUC shown in Fig. 5.2 were used in the life-cycle modeling of rapeseed oil. Section 5.4.3 presents rapeseed oil life-cycle GHG emissions incorporating uncertainty. GHG emission savings when rapeseed oil displaces fossil diesel are shown in appendix (section B).

(a)



	Cropland 4 → Cropland 3								Cropland 4 → Cropland 1							
	CTD HACS	CTD SS	CTM HACS	CTM SS	WTD HACS	WTD SS	WTM HACS	WTM SS	CTD HACS	CTD SS	CTM HACS	CTM SS	WTD HACS	WTD SS	WTM HACS	WTM SS
CV	207	202	232	231	198	204	230	234	49	49	57	56	49	48	57	57

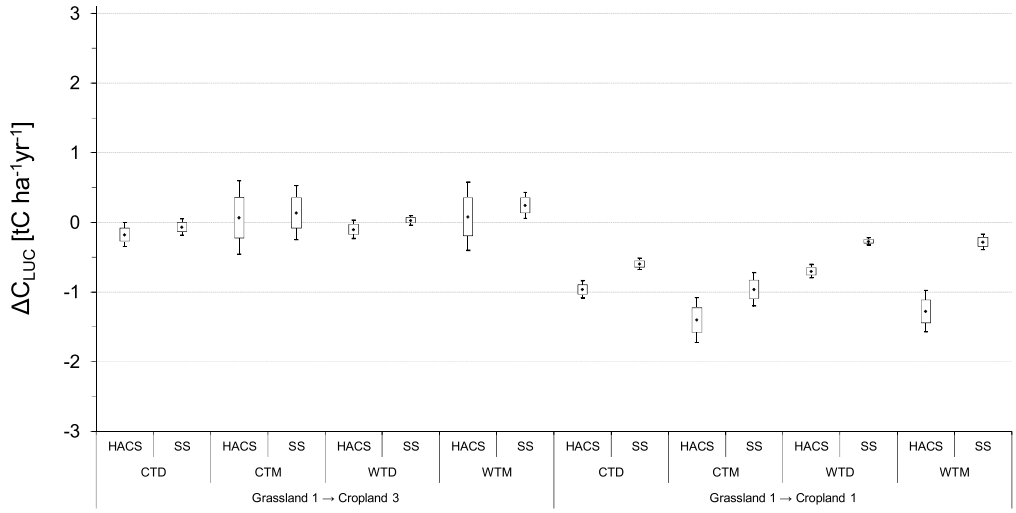
(b)



	Grassland 3 → Cropland 3								Grassland 3 → Cropland 1							
	CTD HACS	CTD SS	CTM HACS	CTM SS	WTD HACS	WTD SS	WTM HACS	WTM SS	CTD HACS	CTD SS	CTM HACS	CTM SS	WTD HACS	WTD SS	WTM HACS	WTM SS
CV	36	33	35	32	34	27	34	26	497	231	122	98	317	104	116	56

Fig. 5.2. Soil carbon exchange associated with the 48 different LUC scenarios assessed (CV: Coefficient of Variation).

(c)



	Grassland 1 → Cropland 3								Grassland 1 → Cropland 1							
	CTD HACS	CTD SS	CTM HACS	CTM SS	WTD HACS	WTD SS	WTM HACS	WTM SS	CTD HACS	CTD SS	CTM HACS	CTM SS	WTD HACS	WTD SS	WTM HACS	WTM SS
CV	61	108	513	186	80	171	396	51	8	9	15	16	9	12	15	25

Fig. 5.2 (cont.). Soil carbon exchange associated with the 48 different LUC scenarios assessed (CV: Coefficient of Variation).

Table 5.5. Selected probability distributions for soil carbon exchange (actual land use: cropland 3; loc: location; sr: selected range).

LUC scenario	Soil Type	Climate Region	Selected probability distributions for ΔC_{LUC} [t C ha^{-1} yr^{-1}]
Cropland 4→ Cropland 3	HACS	CTD	Lognormal (loc: -0.55; μ=0.20; σ=0.40; sr: from -0.39 to 1.11)
		CTM	Maximum Extreme (likeliest: 0.00; scale: 0.67; sr: from -1.17 to 2.41)
		WTD	Lognormal (loc: -0.42; μ=0.15; σ=0.30; sr: from -0.29 to 0.85)
		WTM	Maximum Extreme (likeliest: 0.00; scale: 0.62; sr: from -1.08 to 2.23)
	SS	CTD	Lognormal (loc: -0.37; μ=0.13; σ=0.27; sr: from -0.26 to 0.76)
		CTM	Maximum Extreme (likeliest: 0.00; scale: 0.50; sr: from -0.87 to 1.80)
		WTD	Lognormal (loc: -0.21; μ=0.07; σ=0.15; sr: from -0.15 to 0.42)
		WTM	Maximum Extreme (likeliest: 0.00; scale: 0.24; sr: from -0.42 to 0.86)
Grassland 3 → Cropland 3	HACS	CTD	Gamma (loc: 0.07; scale: 0.16; shape: 4.57; sr: from 0.25 to 1.43)
		CTM	Beta (min: 0.64; max: 3.57; α: 1.61; β: 1.99; sr: from 0.67 to 3.34)
		WTD	Gamma (loc: -0.07; scale: 0.12; shape: 4.57; sr: from 0.07 to 0.96)
		WTM	Beta (min: 0.62; max: 3.33; α: 1.61; β: 1.99; sr: from 0.64 to 3.11)
	SS	CTD	Gamma (loc: -0.06; scale: 0.11; shape: 4.57; sr: from 0.06 to 0.86)
		CTM	Beta (min: 0.57; max: 2.75; α: 1.61; β: 1.99; sr: from 0.59 to 2.58)
		WTD	Gamma (loc: -0.04; scale: 0.06; shape: 4.57; sr: from 0.03 to 0.48)
		WTM	Beta (min: 0.45; max: 1.50; α: 1.61; β: 1.99; sr: from 0.46 to 1.41)
Grassland 1 → Cropland 3	HACS	CTD	Uniform (min: -0.37; max: 0.01)
		CTM	Uniform (min: -0.52; max: 0.65)
		WTD	Uniform (min: -0.25; max: 0.04)
		WTM	Uniform (min: -0.46; max: 0.63)
	SS	CTD	Uniform (min: -0.20; max: 0.06)
		CTM	Uniform (min: -0.30; max: 0.57)
		WTD	Uniform (min: -0.05; max: 0.10)
		WTM	Uniform (min: 0.03; max: 0.45)

Table 5.5 (cont). Selected probability distributions for soil carbon exchange (actual land use: cropland 1; loc: location; sr: selected range).

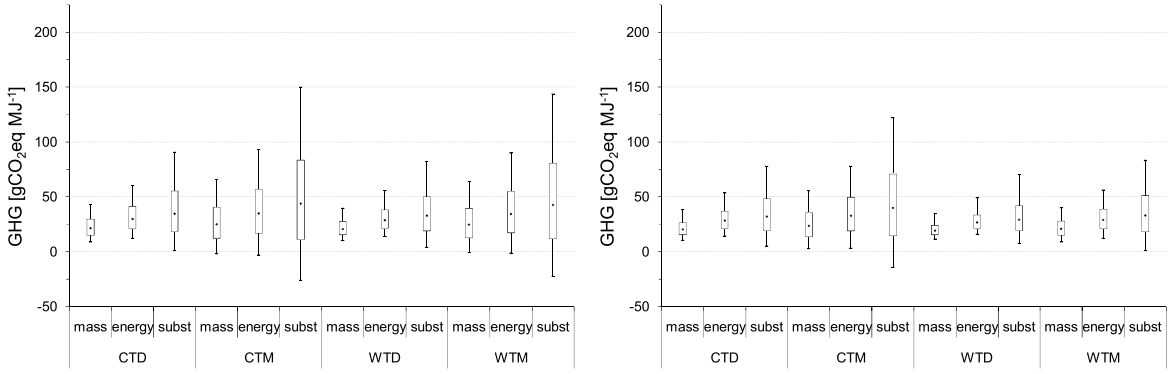
LUC scenario	Soil Type	Climate Region	Selected probability distributions for ΔC_{LUC} [t C ha^{-1} yr^{-1}]
Cropland 4→ Cropland 1	HACS	CTD	Lognormal (loc: -1.31; μ=-0.58; σ=0.41; sr: from -1.11 to 0.27)
		CTM	Lognormal (loc: -3.21; μ=-1.13; σ=0.84; sr: from -2.41 to 0.71)
		WTD	Lognormal (loc: -0.99; μ=-0.44; σ=0.31; sr: from -0.85 to 0.21)
		WTM	Lognormal (loc: -2.97; μ=-1.05; σ=0.78; sr: from -2.23 to 0.66)
	SS	CTD	Lognormal (loc: -0.89; μ=-0.39; σ=0.28; sr: from -0.76 to 0.19)
		CTM	Lognormal (loc: -2.40; μ=-0.85; σ=0.63; sr: from -1.80 to 0.53)
		WTD	Lognormal (loc: -0.50; μ=-0.22; σ=0.16; sr: from -0.42 to 0.10)
		WTM	Lognormal (loc: -1.15; μ=-0.40; σ=0.30; sr: from -0.86 to 0.25)
Grassland 3 → Cropland 1	HACS	CTD	Uniform (min: -0.47; max: 0.59)
		CTM	Uniform (min: -0.58; max: 1.63)
		WTD	Uniform (min: -0.33; max: 0.48)
		WTM	Uniform (min: -0.51; max: 1.54)
	SS	CTD	Uniform (min: -0.27; max: 0.45)
		CTM	Uniform (min: -0.34; max: 1.30)
		WTD	Uniform (min: -0.09; max: 0.32)
		WTM	Uniform (min: 0.01; max: 0.80)
Grassland 1 → Cropland 1	HACS	CTD	Uniform (min: -1.10; max: -0.83)
		CTM	Uniform (min: -1.76; max: -1.05)
		WTD	Uniform (min: -0.81 max: -0.60)
		WTM	Uniform (min: -1.61; max: -0.95)
	SS	CTD	Uniform (min: -0.69; max: -0.51)
		CTM	Uniform (min: -1.23; max: -0.70)
		WTD	Uniform (min: -0.33; max: -0.22)
		WTM	Uniform (min: -0.41; max: -0.16)

5.4.3. GHG intensity results

The GHG intensity of rapeseed oil production in Europe was calculated for two actual land uses – "cropland 1" and "cropland 3" – as depicted in Figs. 5.3 and 5.4, respectively. Figure 5.3 shows GHG emissions for 24 scenarios (3 reference land uses, 4 climate regions and 2 types of soil) addressing the main management practices and levels of carbon input for rapeseed cultivation with the exception of manure application (actual land use "cropland 3" as described in section 5.4.2). Figure 5.4 shows GHG emissions for the remaining 24 scenarios, in which rapeseed cultivation practices include manure application ("cropland 1" according to section 5.4.2).

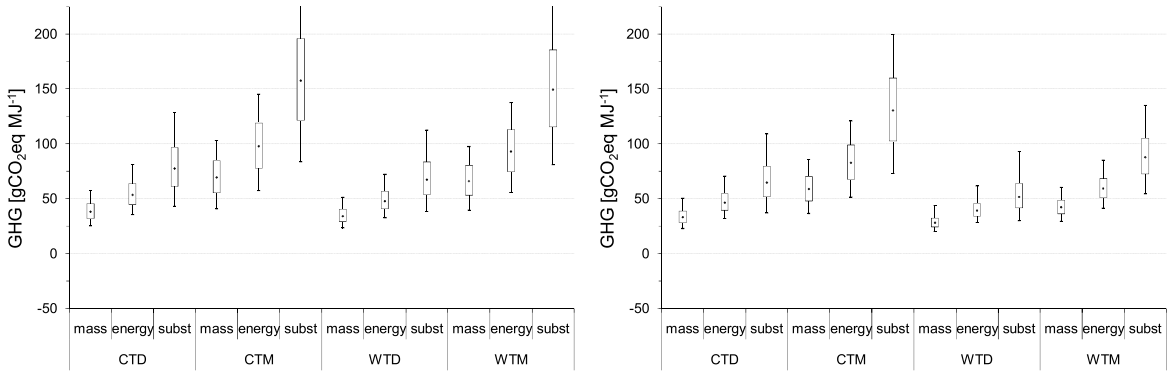
The technique of Monte Carlo simulation has been used for uncertainty propagation, with

a random sampling procedure and 30000 iterations per simulation. The output distributions are divided in the 5th, 25th, 50th, 75th, and 95th percentiles and displayed in box plots. A thorough explanation of the results follows, focusing firstly on median values and secondly on uncertainty ranges (parametric uncertainty).



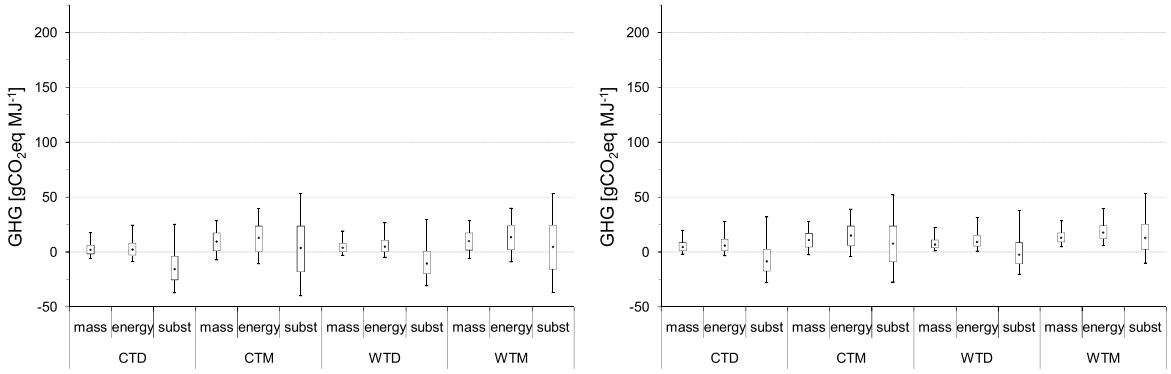
Cropland 4 → Cropland 3, HACS

Cropland 4 → Cropland 3, SS



Grassland 3 → Cropland 3, HACS

Grassland 3 → Cropland 3, SS



Grassland 1 → Cropland 3, HACS

Grassland 1 → Cropland 3, SS

Fig. 5.3. Rapeseed Oil life-cycle GHG emission results (actual land use: "cropland 3"). HACS: High activity clay soils; SS: Sandy soils.

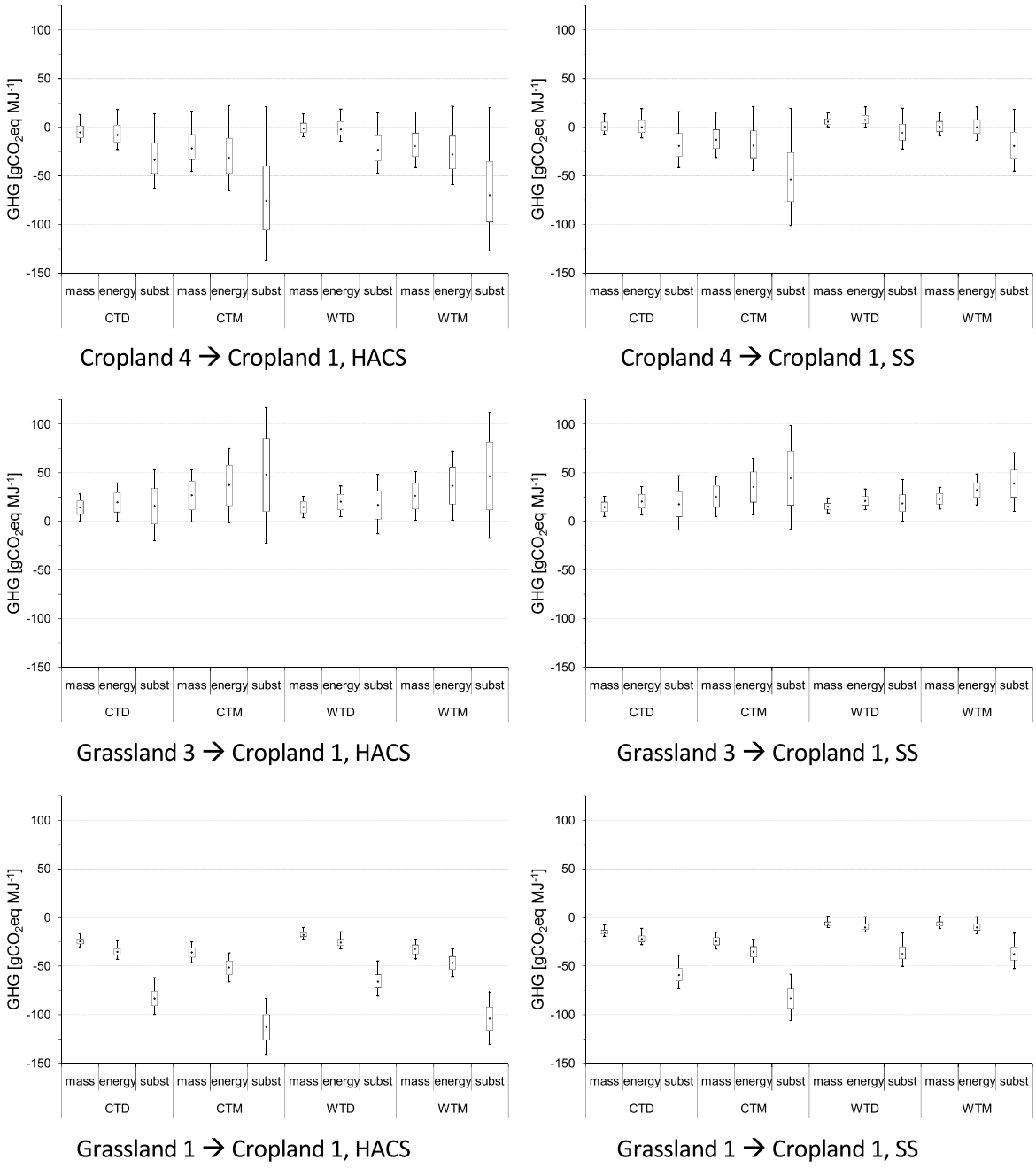


Fig. 5.4. Rapeseed Oil life-cycle GHG emission results (actual land use: "cropland 1"). HACS: High activity clay soils; SS: Sandy soils.

Comparing Figs. 5.3 and 5.4, it is shown that median values in Fig. 5.4 are considerably lower for the same scenarios, with negative emissions in several situations. This can be

explained by the actual land use considered in Fig. 5.4 (“cropland 1”). “Cropland 1” denotes cropland in which manure is used as an input in soil management. As a result, highest levels of carbon fixation are achieved in the soil according to EC (2010a) data. Moreover, lower levels of mineral fertilization are required, with corresponding lower life-cycle GHG emissions due to avoided fertilizer production.

Within the same typology of land use change (“grassland” to “cropland”), significantly different GHG intensities can be registered: “grassland 1 to cropland 3” in Fig. 5.3 has lower emissions than “grassland 3 to cropland 3”, because the reference land use “grassland 1” corresponds to severely degraded grassland which has a lower carbon stock according to EC (2010a) data. Furthermore, land use change occurring in degraded land is credited with a 29 g $CO_{2eq}MJ^{-1}$ bonus (EC 2010a), which lowers further the LUC emissions in this set of scenarios. An important conclusion is that in addition to the type of land use change considered, the agricultural practices adopted also have a significant impact in the GHG intensity of rapeseed oil. Taking “cropland 4 to cropland 3” as a reference scenario in Fig. 5.3, it is shown that the “grassland 3 to cropland 3” scenario has higher GHG emissions, whereas the “grassland 1 to cropland 3” scenario presents lower GHG emissions. The divergence lies in the different management and input factors already stated for the agricultural practices “grassland 3” and “grassland 1”, which result in distinct soil carbon stock variations. The same conclusion can be drawn from Fig. 5.4.

The set of scenarios with “grassland 3” as the reference land use show the highest GHG intensity in both Figs. 5.3 and 5.4, clearly deviating from the remaining two sets (“cropland 4” and “grassland 1” reference land uses). This can be explained by the high soil carbon stock associated with the reference land use “grassland 3”, which is released to the atmosphere as a consequence of land use conversion. Particularly high levels of GHG emissions are calculated for moist climates, irrespective of soil type, because the standard soil organic carbon SOC_{ST} in this case is higher than in the case of dry climates (see Table 5.1). Moreover, actual land uses “cropland 3” and “cropland 1” cannot accumulate soil carbon stocks in moist climates as high as in dry climates. Maximum GHG emissions are calculated in the land use conversion from “grassland 3” to “cropland 3” (in a HACS soil,

within a cold temperate moist region), with median values that can be significantly high and scattered, ranging approximately from 70 to 150 g $CO_{2eq}MJ^{-1}$.

Performing allocation has a damping effect on calculations, because emissions are split between co-products based on a specific relationship. Net GHG emissions are reduced when allocation is performed, which is advantageous for the GHG balance of the final biofuel product; conversely, allocation diminishes the GHG benefits of the biofuel if GHG sequestration (i.e. negative GHG emissions) is calculated over the life-cycle. The substitution method, on the other hand, always subtracts from the biofuel chain the credits associated with displaced products, meaning that it always works in the same direction for a specific chain. Moreover, this method does not introduce the artificial damping effect associated with the allocation approach.

The difference between allocation and substitution motivates that whenever GHG emissions are positive but relatively low, the lower damping effect of allocation (in absolute terms) and the credits from the substitution method tend to approximate the results. This is shown in Fig. 5.3, where the sets of scenarios “cropland 4 to cropland 3” and “grassland 1 to cropland 3” have median GHG emissions close to the corresponding averages (31 and 3 g $CO_{2eq}MJ^{-1}$, respectively). On the other hand, when GHG emissions are negative, the calculated values with the substitution method deviate markedly from the allocation results, as shown in Fig. 5.4.

Concerning parametric uncertainty, Figs. 5.3 and 5.4 show probability distributions for rapeseed oil GHG intensity. Focusing on the 25^{th} and 75^{th} percentiles (top and bottom of the boxes), it is shown that the uncertainty ranges with energy allocation and the substitution method are higher than the uncertainty ranges using mass allocation. In the former, this can be explained by the higher LHV of rapeseed oil in comparison to rape meal, which results in a high level of emissions allocated to rapeseed oil. In the latter, the higher uncertainty range is due to the additional uncertainty introduced by the displaced product: credits for soy meal substitution by rape meal bring in the uncertainty associated with soy meal production. Moreover, the substitution method does not damp the uncertainty range, as already explained for median values.

The uncertainty level in the first two sets of scenarios ("cropland 4 to cropland 3" and "grassland 3 to cropland 3") is higher than the uncertainty in "grassland 1 to cropland 3", which is explained by the deterministic assessment of the reference land use "grassland 1". Furthermore, in several situations in Figs. 5.3 and 5.4 parameter uncertainty ranges clearly overwhelm differences between average values of specific scenarios. This calls for the need of including parameter uncertainty in the life-cycle assessment of biofuel systems, as already emphasized in chapter 2, in order to avoid the sometimes erroneous impression of distinguishability among alternative scenarios.

Comparing current soil types for rapeseed cultivation in Europe, Figs. 5.3 and 5.4 show that no significant difference exists in terms of RO GHG intensity between high activity clay soils (HACS) and sandy soils (SS). The exception is in warm temperate moist (WTM) regions, where the standard carbon content of those soils is very different (see Table 5.1), which may result in significant differences in median values and uncertainty ranges.

The contribution of input data to the variance of rapeseed oil GHG emissions is listed in Tables 5.6 and 5.7 for the two actual land uses considered ("cropland 3" and "cropland 1"). A high contribution of soil C emissions is shown: in the majority of the assessed scenarios, land use change and agricultural practices are the most important source of variability of results, overriding the contribution of other parameters, namely soil N_2O emissions. This is particularly important, since in the review of biofuel life-cycle studies presented in chapter 3 only a few considered the contribution of soil carbon emissions in the GHG balance (Malça and Freire 2011a). Other authors also raise these concerns (de Vries et al. 2010; van der Voet et al. 2010).

As opposed to the deterministic approaches of (i) ignoring soil carbon emissions from land use change or (ii) simply using single factors to characterize the soil carbon content of each land category (cropland, grassland), this section shows that taking into account agricultural practices, namely different management and input options, may introduce significant levels of uncertainty in the life-cycle GHG balance of a biofuel system. This happens to be the case even within the same land use in the reference and actual scenarios (e.g. cropland to cropland). Moreover, the large uncertainties computed for specific scenarios of LUC and the resulting significant impact of soil carbon emissions in the GHG intensity of biofuels

demonstrate that a detailed assessment incorporating uncertainty is required to justify in what conditions the promotion of first-generation biofuels as GHG savers over petroleum fuels is actually the best option.

Table 5.6. Contribution of input data to the variance of RO GHG emissions (in %). Actual land use: Cropland 3. n/a: not applicable.

LUC scenario	Soil Type	Parameter	CTD		CTM		WTD		WTM	
			energy	subst.	energy	subst.	energy	subst.	energy	subst.
Cropland 4 → Cropland 3	HACS	Soil carbon emissions	69.7	66.6	91.7	90.8	57.7	54.4	90.2	89.1
		Soil N_2O emissions	21.7	21.1	5.7	5.6	30.1	26.7	7.0	6.9
		N_2O GWP	4.0	3.8	1.2	1.2	5.5	5.0	1.2	1.2
		Soy meal substitution credit	n/a	4.0	n/a	1.0	n/a	5.7	n/a	1.1
		TOTAL	*95.4*	*95.5*	*98.6*	*98.6*	*93.3*	*91.8*	*98.4*	*98.3*
	SS	Soil carbon emissions	52.4	48.6	86.5	85.3	27.7	24.1	62.2	58.8
		Soil N_2O emissions	33.8	31.6	9.6	9.5	50.1	46.0	27.1	25.7
		N_2O GWP	6.4	6.0	1.8	1.8	10.3	9.1	4.9	4.6
		Soy meal substitution credit	n/a	6.8	n/a	1.4	n/a	10.4	n/a	5.3
		N fertilizer production	-	-	-	-	4.5	4.0	-	-
		TOTAL	*92.6*	*93.0*	*97.9*	*98.0*	*92.6*	*93.6*	*94.2*	*94.4*
Grassland 3 → Cropland 3	HACS	Soil carbon emissions	63.0	59.9	88.3	87.6	52.7	49.4	86.2	85.3
		Soil N_2O emissions	23.9	23.1	6.2	6.2	31.2	29.7	6.9	6.8
		N_2O GWP	4.0	3.8	-	-	5.8	5.5	-	-
		Soy meal substitution credit	n/a	4.7	n/a	0.8	n/a	5.8	n/a	1.1
		Rapeseed yield	-3.6	-3.4	-2.6	-2.7	-3.8	-3.6	-3.1	-3.0
		TOTAL	*94.5*	*94.9*	*97.1*	*97.3*	*93.5*	*94.0*	*96.2*	*96.2*
	SS	Soil carbon emissions	46.6	42.9	82.3	81.0	23.7	20.9	57.3	54.0
		Soil N_2O emissions	34.6	33.1	9.4	9.3	49.8	45.4	26.0	25.0
		N_2O GWP	6.8	6.3	2.1	2.1	10.2	9.2	5.0	4.7
		Soy meal substitution credit	n/a	6.6	n/a	1.6	n/a	10.3	n/a	5.2
		Rapeseed yield	-4.1	-3.8	-3.2	-3.1	-4.2	-3.6	-5.3	-5.0
		N fertilizer production	-	-	-	-	4.5	4.0	-	-
		N fertilizer application rate	-	-	-	-	3.9	3.5	-	-
		TOTAL	*92.1*	*92.7*	*97.0*	*97.1*	*96.3*	*96.9*	*93.6*	*93.9*
Grassland 1 → Cropland 3	HACS	Soil carbon emissions	28.8	24.4	75.4	73.1	18.6	15.5	72.0	69.5
		Soil N_2O emissions	51.2	46.2	17.3	16.9	56.7	51.0	20.4	19.8
		N_2O GWP	10.3	9.3	3.5	3.3	12.4	11.0	3.5	3.4
		Soy meal substitution credit	n/a	11.5	n/a	3.1	n/a	12.1	n/a	3.4
		N fertilizer production	3.9	3.3	-	-	4.7	3.9	-	-
		N fertilizer application rate	3.6	3.2	-	-	4.6	4.0	-	-
		TOTAL	*97.8*	*97.9*	*96.2*	*96.4*	*97.0*	*97.5*	*95.9*	*96.1*
	SS	Soil carbon emissions	15.5	12.6	63.8	60.5	6.0	4.8	31.8	27.8
		Soil N_2O emissions	59.2	52.9	26.0	25.0	62.9	55.0	46.5	43.1
		N_2O GWP	12.6	10.8	4.7	4.5	14.4	11.8	9.2	8.2
		Soy meal substitution credit	n/a	13.1	n/a	4.8	n/a	14.6	n/a	9.8
		N fertilizer production	4.9	4.0	-	-	5.6	4.6	3.5	3.1
		N fertilizer application rate	4.7	3.9	-	-	5.4	4.6	3.6	3.3
		TOTAL	*96.9*	*97.3*	*94.5*	*94.8*	*94.3*	*95.4*	*94.6*	*95.3*

Table 5.7. Contribution of input data to the variance of RO GHG emissions (in %). Actual land use: Cropland 1. n/a: not applicable.

LUC scenario	Soil Type	Parameter	CTD		CTM		WTD		WTM	
			energy	subst.	energy	subst.	energy	subst.	energy	subst.
Cropland 4 → Cropland 1	HACS	Soil carbon emissions	88.4	83.4	96.5	95.4	81.7	74.6	96.3	95.0
		Soil N_2O emissions	8.3	8.0	1.8	1.8	12.9	12.1	2.2	2.1
		N_2O GWP	-	-	-	-	2.5	-	-	-
		Soy meal substitution credit	n/a	5.6	n/a	1.1	n/a	8.5	n/a	1.4
		TOTAL	*96.7*	*97.0*	*98.3*	*98.3*	*97.1*	*95.2*	*98.5*	*98.5*
	SS	Soil carbon emissions	79.5	71.2	94.6	92.3	56.3	44.1	83.7	77.3
		Soil N_2O emissions	14.5	13.2	3.5	3.5	31.3	25.7	12.4	11.7
		N_2O GWP	3.0	2.7	-	-	6.6	5.3	-	-
		Soy meal substitution credit	n/a	10.3	n/a	2.4	n/a	20.5	n/a	7.5
		TOTAL	*97.0*	*97.4*	*98.1*	*98.2*	*94.2*	*95.6*	*96.1*	*96.5*
Grassland 3 → Cropland 1	HACS	Soil carbon emissions	89.7	85.4	96.8	95.7	83.3	76.6	96.0	94.8
		Soil N_2O emissions	7.3	6.9	1.9	1.8	11.1	10.3	2.2	2.1
		N_2O GWP	-	-	-	-	-	-	-	-
		Soy meal substitution credit	n/a	4.9	n/a	1.2	n/a	8.0	n/a	1.3
		TOTAL	*97.0*	*97.2*	*98.7*	*98.7*	*94.4*	*94.9*	*98.2*	*98.2*
	SS	Soil carbon emissions	81.2	74.3	94.3	92.1	59.3	47.2	96.0	94.8
		Soil N_2O emissions	12.7	11.9	3.3	3.2	27.5	23.7	2.2	2.1
		N_2O GWP	-	-	-	-	5.3	4.3	-	-
		Soy meal substitution credit	n/a	8.4	n/a	2.3	n/a	18.3	n/a	1.3
		TOTAL	*93.9*	*94.6*	*97.6*	*97.6*	*92.1*	*93.5*	*98.2*	*98.2*
Grassland 1 → Cropland 1	HACS	Soil carbon emissions	37.6	27.3	76.2	69.2	28.9	19.2	75.1	67.6
		Soil N_2O emissions	39.5	30.7	12.2	11.3	46.7	34.5	18.9	12.5
		N_2O GWP	7.5	5.8	2.8	2.5	10.0	7.2	3.3	2.8
		Soy meal substitution credit	n/a	24.6	n/a	9.0	n/a	29.2	n/a	9.9
		Rapeseed yield	5.5	3.6	4.6	4.1	3.1	-	3.9	3.6
		N fertilizer application rate	3.1	2.4	-	-	3.8	2.9	-	-
		N fertilizer production	-	-	-	-	3.6	-	-	-
		TOTAL	*93.2*	*94.4*	*95.8*	*96.1*	*96.1*	*93.0*	*96.2*	*96.4*
	SS	Soil carbon emissions	25.2	15.6	69.3	59.5	11.7	6.3	39.1	25.9
		Soil N_2O emissions	51.0	36.0	19.6	17.5	59.6	40.4	43.0	32.9
		N_2O GWP	11.0	7.6	3.8	3.5	15.6	8.1	8.6	6.3
		Soy meal substitution credit	n/a	32.7	n/a	12.9	n/a	35.4	n/a	28.4
		N fertilizer production	3.8	-	-	-	5.2	2.9	3.4	-
		N fertilizer application rate	3.5	-	-	-	4.9	-	3.2	-
		Rapeseed yield	-	-	-	2.4	-	-	-	-
		Diesel fuel agricultural machinery	-	-	-	-	3.8	-	-	-
		TOTAL	*94.5*	*91.9*	*92.7*	*93.4*	*98.8*	*93.1*	*97.3*	*93.5*

5.5. dLUC MODELING: DETERMINISTIC VERSUS HYBRID APPROACH

In this section, two alternative approaches for estimating the variation in soil carbon content associated with land use change are assessed. The implications of these approaches in the GHG intensity of biofuel systems are also analyzed. The calculation method for ΔC_{LUC} is followed (i) firstly by assigning deterministic coefficients for land use types and agricultural practices (management and input)[7] as indicated in EC (2010a); and (ii) secondly by combining deterministic coefficients of EC (2010a) with the error ranges proposed in IPCC (2006) (hybrid approach).

5.5.1. Land use change scenarios

The assessment is conducted for five different scenarios of land use change, as shown in Table 5.8, and is exemplified with wheat cultivation: the actual land use for all scenarios is wheat cultivation, with reduced tillage and medium inputs (label A). In terms of reference land use, the categories "grassland" and "cropland" were each divided into representative situations of extreme variations in soil carbon stocks (ΔC_{LUC}). Finally, a fifth baseline scenario is considered in which the agricultural practices are maintained between reference and actual land uses (C → A). Results presented in this section assume that wheat cultivation occurs in high activity clay soils (HACS) in a cool temperate moist (CTM) climate, but the effects on the biofuel GHG intensity of other soil types and climate regions are discussed in the sensitivity analysis of chapter 6.

Table 5.8. LU and LUC scenarios considered in the analysis of section 5.5.

Land use scenarios	Label	LUC scenarios considered
Grassland, improved	*Gi*	*Gi → A (GiA)*
Grassland, severely degraded	*Gd*	*Gd → A (GdA)*
Cropland, full tillage, low inputs	Cf	*Cf → A (CfA)*
Cropland, no tillage, high with manure	*Cm*	*Cm → A (CmA)*
Cropland, reduced tillage, medium inputs	*C*	*C → A (CA)*

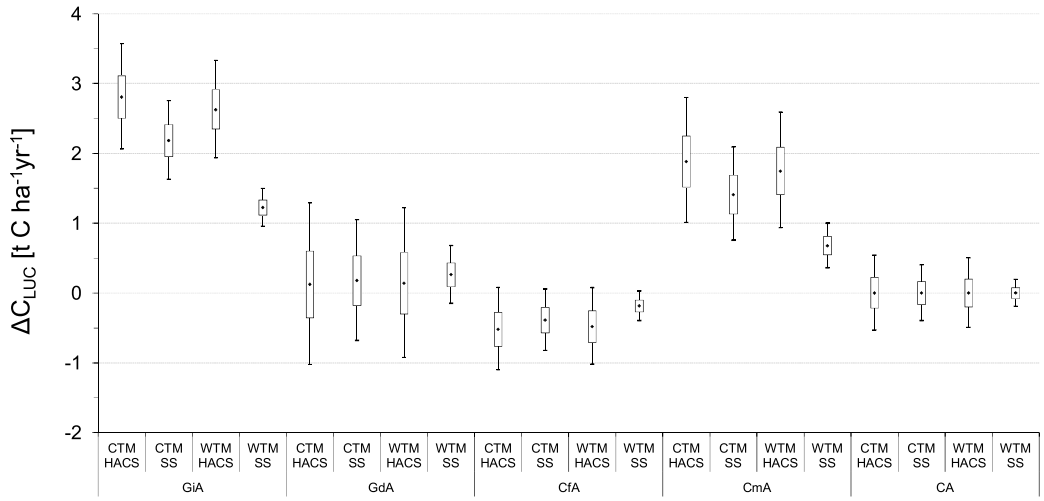
[7] These coefficients draw on the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006).

5.5.2. Probability distributions for soil carbon exchange

Fig. 5.5 shows calculated variations in land carbon stocks (ΔC_{LUC}) for the five LUC scenarios under analysis using the hybrid approach. Probability distributions of ΔC_{LUC} are listed in Table 5.9. Representative climate regions (CTM and WTM) and soil types (HACS and SS) for wheat cultivation in Europe were selected. Each ΔC_{LUC} probability distribution has been computed as follows:

- firstly, the coefficients F_{LU}, F_{MG}, and F_I for each land use scenario were assigned probability distributions based on the error ranges indicated in IPCC (2006);
- secondly, annualized soil carbon stock variations due to LUC were computed (Monte-Carlo simulation) using the calculation rules of EPC (2009) and EC (2010a), as explained in chapter 2 (equations 2.8 to 2.10).

Fig. 5.5 shows that the uncertainty of ΔC_{LUC} when the error ranges of IPCC (2006) guidelines are taken into account is very high. This high level of uncertainty can be compared with calculated values when EC (2010a) deterministic data is considered (Table 5.10). Two scenarios result in the highest soil carbon emissions from land use change, namely GiA and CmA. In the first case, a carbon-rich soil (improved grassland Gi) is converted, whereas in the second the combination of no tillage and manure application (Cm) also provides a high level of soil carbon content. Both scenarios are thus net emitters of soil carbon due to land use conversion to wheat cultivation. Conversely, the CfA scenario reaches the maximum levels of carbon sequestration due to LUC, because the reference land use considered (full tillage cultivation with low inputs to soil) is associated to minimum levels of soil carbon content.



	GiA				GdA				CfA				CmA				CA			
	CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS
CV	16	15	16	13	568	296	467	96	70	70	70	70	29	29	29	29	–	–	–	–

Fig. 5.5. Calculated soil carbon exchange values using the hybrid approach (CV: coefficient of variation).

Table 5.9. Selected probability distributions for soil carbon exchange using the hybrid approach.

LUC scenario	Climate Region	Soil Type	Selected probability distributions for ΔC_{LUC} [t C ha^{-1} yr^{-1}]
CA	CTM	HACS	Normal (μ=0.00; σ=0.33)
		SS	Normal (μ=0.00; σ=0.25)
	WTM	HACS	Lognormal (location: -36.68; μ=0.00; σ=0.30)
		SS	Normal (μ=0.00; σ=0.12)
GiA	CTM	HACS	Lognormal (location: -27.55; μ=2.82; σ=0.45)
		SS	Lognormal (location: -57.73; μ=2.19; σ=0.34)
	WTM	HACS	Lognormal (location: -24.22; μ=2.63; σ=0.42)
		SS	Lognormal (location: -13.14; μ=1.23; σ=0.16)
GdA	CTM	HACS	Lognormal (location: -85.48; μ=0.13; σ=0.71)
		SS	Lognormal (location: -95.50; μ=0.18; σ=0.53)
	WTM	HACS	Lognormal (location: -242.34; μ=0.14; σ=0.66)
		SS	Beta (min: -2.71; max: 3.34; α: 70.26; β: 72.66)
CfA	CTM	HACS	Lognormal (location: -31.89; μ=-0.52; σ=0.36)
		SS	Lognormal (location: -20.96; μ=-0.39; σ=0.27)
	WTM	HACS	Lognormal (location: -18.71; μ=-0.48; σ=0.33)
		SS	Lognormal (location: -7.03; μ=-0.19; σ=0.13)
CmA	CTM	HACS	Lognormal (location: -14.45; μ=1.89; σ=0.55)
		SS	Gamma (location: -4.95; scale: 0.03; shape: 243.83)
	WTM	HACS	Beta (min: -1.85; max: 7.84; α: 31.88; β: 53.94)
		SS	Lognormal (location: -5.16; μ=0.67; σ=0.20)

Table 5.10. Calculated ΔC_{LUC} [t C ha^{-1} yr^{-1}] using the deterministic coefficients of EC (2010a).

GiA				GdA				CfA				CmA				CA			
CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS	CTM HACS	CTM SS	WTM HACS	WTM SS
2,811	2,187	2,629	1,224	0,125	0,180	0,141	0,263	-0,524	-0,392	-0,486	-0,188	1,888	1,411	1,749	0,676	0,000	0,000	0,000	0,000

5.5.3. GHG intensity results

In this section, the GHG intensity of bioethanol from wheat is calculated for the five LUC scenarios described in the previous section (GiA, GdA, CfA, CmA, CA). To emphasize the importance of land use change in the life-cycle GHG intensity of wheat bioethanol (the illustrative example in this section), Fig. 5.6 shows ΔC_{LUC} values disaggregated from the rest of the life-cycle.

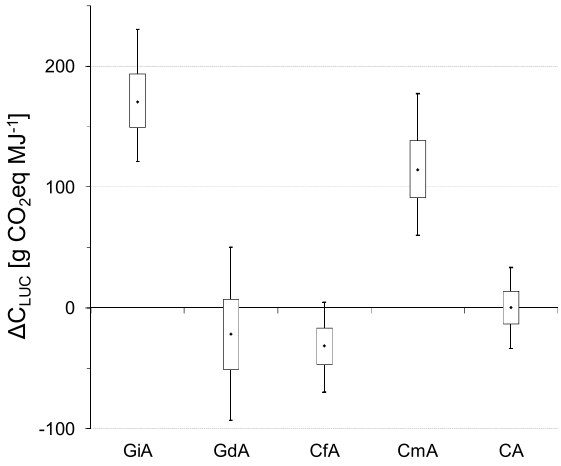


Fig. 5.6. Calculated soil carbon exchange values (in g CO_2eq MJ^{-1}) using the hybrid approach (type of soil: HACS; climate region: CTM). Uncertainty in wheat yield and bioethanol production has been taken into account.

Fig. 5.7 shows the GHG intensity of bioethanol from wheat using different approaches for dealing with co-products: mass, energy and economic allocation methods, a

substitution method and the "no allocation" approach. Left column of Fig. 5.7 shows GHG intensity results calculated with deterministic coefficients from EC (2010a), whereas GHG results with error ranges from IPCC (2006) are shown on the right column. It should be noted that:

- the "CA (a)" results of Fig. 5.7(a) are also results with no net carbon emissions from land use change (i.e. ΔC_{LUC} = 0 t C ha^{-1} yr^{-1}), since the reference and actual land uses are the same in this scenario and only deterministic coefficients from EC (2010a) are used for the calculation of ΔC_{LUC};
- even using deterministic values for ΔC_{LUC}, all graphs on the left of Fig. 5.7 show uncertainty ranges because all other uncertain parameters in the wheat-based bioethanol model retain their uncertainty ranges.

Firstly, a comparison of ΔC_{LUC} values in Fig.5.6 with non-allocated life-cycle results in Fig.5.7b shows the importance of LUC in the GHG assessment of biofuels, both in terms of average values and uncertainty ranges. Secondly, comparing the deterministic and hybrid approaches, Fig. 5.7 shows that the inclusion of IPCC (2006) error ranges in the calculations has a significant influence on the results, as demonstrated by the wider range for GHG intensity of the right-hand column of Fig. 5.7. This is particularly evident for the GdA land use change scenario, because this case has the higher uncertainty: according to IPCC (2006), there is an error range of ± 40% in the soil carbon stock factor for grassland management in severely degraded land.

Although average values do not change with the inclusion of (symmetric) error ranges from IPCC (2006), it should be noted that taking into account this additional source of uncertainty in the CA LUC scenario [CA(b) in Fig. 5.7] eliminates the differences between co-product scenarios previously shown [CA(a), left side of Fig. 5.7]. Actually, what at first seemed a distinction between different co-product approaches became much less evident after incorporating IPCC (2006) uncertainty data.

Fig. 5.7 demonstrates the implications in the GHG intensity results of a biofuel system of including the IPCC (2006) uncertainty in soil carbon stock factors. It is therefore

important that this type of uncertainty be included in the GHG intensity and GHG emission saving calculations of biofuel systems.

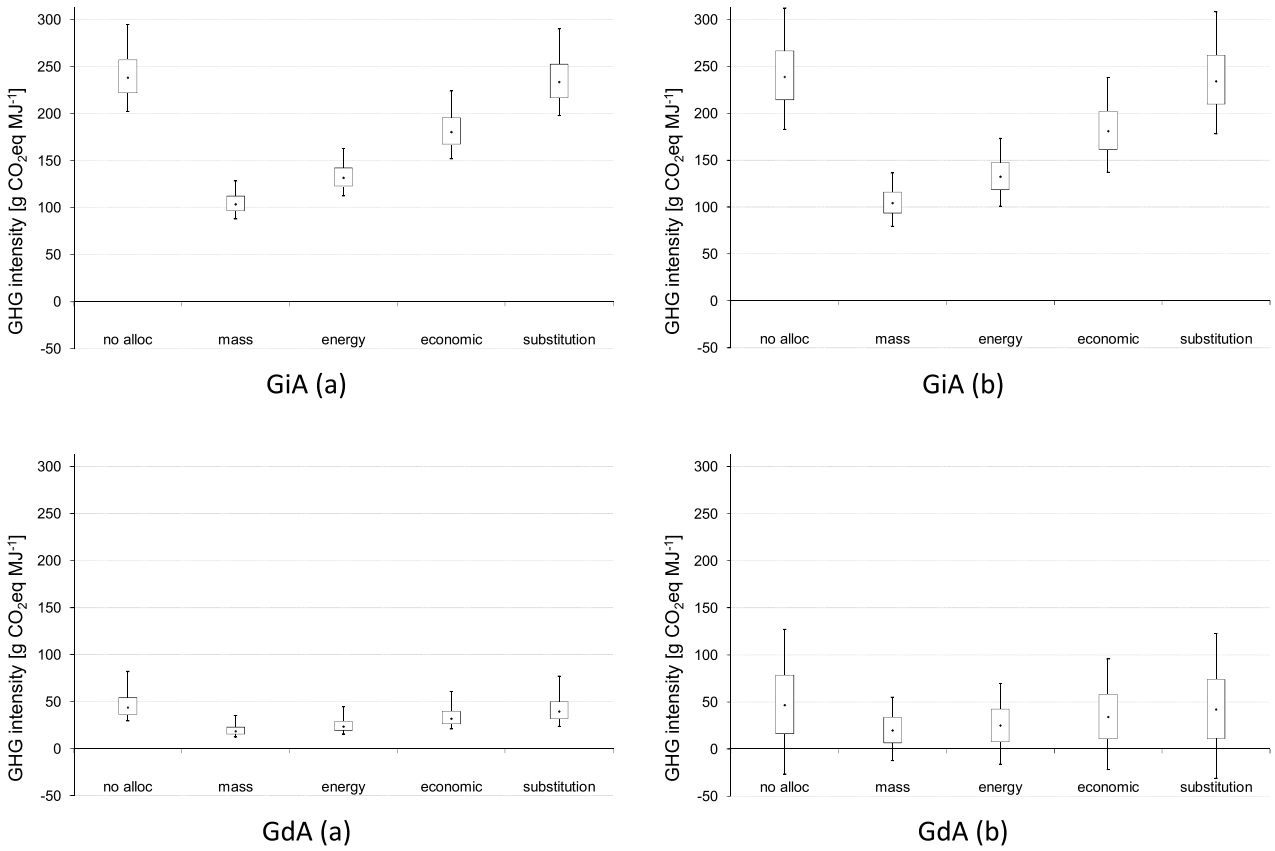


Fig. 5.7. Life-cycle GHG intensity for wheat-based bioethanol based on ΔC_{LUC} values calculated from (a) deterministic EC (2010a) coefficients, and (b) the hybrid approach (type of soil: HACS; climate region: CTM).

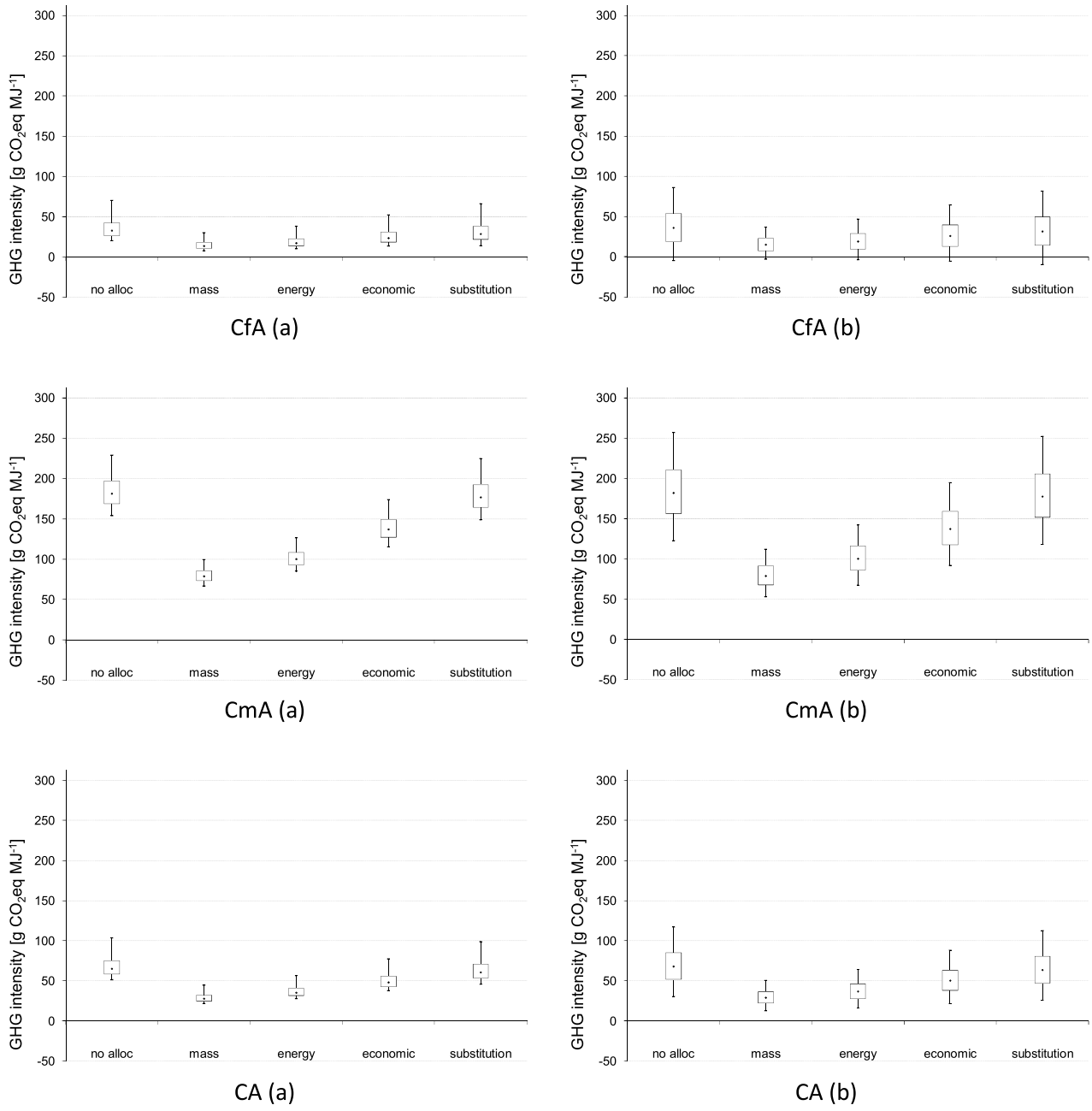


Fig. 5.7 (cont). Life-cycle GHG intensity for wheat-based bioethanol based on ΔC_{LUC} values calculated from (a) deterministic EC (2010a) coefficients, and (b) the hybrid approach (type of soil: HACS; climate region: CTM).

5.6. CONCLUDING REMARKS

This chapter analyses three different approaches for estimating soil carbon fluxes associated with land use change, based on IPCC (2006) guidelines, European legislation (EPC 2009; EC 2010a) and literature data. Calculation of soil carbon exchange values for different land use change scenarios incorporates several sources of uncertainty, namely different agricultural practices (soil management and carbon inputs to soil), climate, and type of soil. Even though the first approach – which uses data at a higher aggregation level – can predict global trends associated with generic LUC scenarios, it lacks the possibility of distinguishing between soil and climate types, as well as the increased refinement level of LUC scenarios provided by the other two approaches. The second approach uses EC (2010a) data and has the merit of evaluating the overall uncertainty within each specific land use change scenario by taking into account all potential agricultural practices. Moreover, a distinction is made between soil and climate types. Finally, the third approach combines EC (2010a) data with error ranges provided in the IPCC (2006) guidelines, which increases significantly the uncertainty range of calculated ΔC_{LUC}. This is the most robust method to address soil carbon exchange due to land use change incorporating uncertainty. Results presented in chapter 6 use this approach.

To conclude, variation of ΔC_{LUC} among the approaches and scenarios addressed in this chapter shows that modeling soil carbon exchange due to LUC is a central aspect in the GHG assessment of energy crops. Moreover, selected examples drawing on European biofuel chains – rapeseed oil and wheat-based bioethanol – emphasize the importance of taking into account ΔC_{LUC} in the life-cycle GHG assessment of biofuel systems.

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6. Main Results and Discussion

"To aid in enabling reliable quantitative uncertainty analysis, the LCA community should develop a better understanding of the importance of different types of uncertainty and variability and develop protocols for reliably characterizing, propagating, and analyzing uncertainty in LCA."

Lloyd and Ries (2007)

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6. MAIN RESULTS AND DISCUSSION

6.1. PURPOSE AND SCOPE

Main results are presented and discussed in this chapter. Each section covers a specific topic with respect to the biofuel systems addressed in this dissertation: RO; RME; wheat bioethanol and bioETBE; and sugar beet bioethanol and bioETBE. With this approach, comparison between biofuel chains is facilitated. Firstly, the energy requirement E_{req} and GHG emissions per life-cycle stage are presented. Secondly, life-cycle energy renewability efficiency and GHG intensity results are calculated for selected cultivation scenarios: improved grassland to cropland (GiA); cropland to cropland CA; and full-tillage cropland to cropland (CfA). An analysis of the contribution of each parameter to the variance of results is also conducted. Thirdly, biofuel systems are compared in terms of land use efficiency. Fourthly, the "carbon payback time" associated to biofuel production displacing petroleum fuel production is calculated. Finally, a sensitivity analysis to important parameters is presented.

LAND USE CHANGE SCENARIOS. Chapter 5 presented several approaches to model land use change and showed its relevance in the life-cycle of biofuels. In chapter 6, three scenarios are selected concerning soil carbon exchange. Three alternative reference land

uses – (i) high input improved grassland (denoted by "Gi"); (ii) medium input, reduced tillage cropland (denoted by "C"); and (iii) low input full-tillage cropland (denoted by "Cf") – are converted into reduced-tillage cropland with medium level of inputs (actual land use, denoted by "A"). Two of these LUC scenarios (GiA and CfA) work as extreme cases of LUC, the former with net C emissions and the latter contributing to C sequestration. The third scenario (CA) works as a conservative scenario, since cultivation management practices remain unchanged between the reference and actual land uses. A baseline scenario with high activity clay soil (HACS) and cool temperate moist (CTM) climate has been selected as the most representative of biofuel production in Europe. Fig. 6.1 shows the ranges for the exchange of soil carbon stocks according to the LUC scenarios and conditions considered (cf. Fig. 5.5).

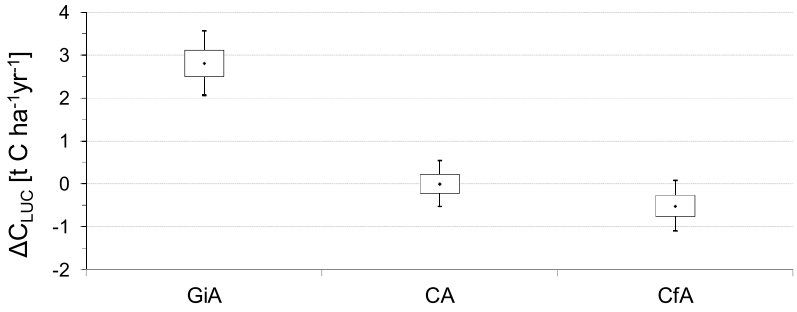


Fig. 6.1. Soil carbon exchange associated with selected LUC scenarios: (i) high-input improved grassland to cropland (GiA); (ii) low-tillage, medium-input cropland to cropland (CA); and (iii) full-tillage, low-input cropland to cropland (CfA). The boxes show the interquartile range, the mark is the median and the ends of the whiskers are the 5th and 95th percentiles. Same notation is used in the remaining figures of the chapter.

6.2. ENERGY REQUIREMENT AND GHG EMISSIONS PER LIFE-CYCLE STAGE

This section presents the energy requirement and GHG intensity of biofuel systems per life-cycle stage, emphasizing the importance of specific stages in the average and/or uncertainty range of results. The following sub-division is considered: land use change; cultivation; first industrial conversion step; second industrial conversion step; and transportation activities. Results by life-cycle stage are non-allocated, i.e. the

contribution of co-products to the chain is not taken into account. Implications of multifunctionality are dealt with in the remaining sections of the chapter. Concerning GHG results, LUC emissions are disaggregated from the rest of the life-cycle, which enables a better understanding of their importance in the overall GHG balance. Results are calculated per MJ of biofuel produced. Therefore, generic soil carbon exchange data presented in Fig. 6.1 has been recalculated to "g CO_2eq MJ^{-1}", and thus takes into account the (uncertain) energy productivity of each crop (MJ ha^{-1}).

The energy requirement E_{req} and GHG emissions of the biofuel systems investigated in this dissertation are shown per life-cycle stage in Figures 6.2 to 6.9[1]. Regardless of the biofuel chain considered, several generic conclusions can be drawn:

- a comparison between (total) E_{req} and GHG emission values shows that uncertainty ranges are higher in the latter;
- land use change has a strong contribution both to the average and uncertainty range of total GHG emissions. Differences between LUC scenarios are clearly retained in total GHG results;
- when land use change is excluded from the analysis, the uncertainty in total GHG emissions comes mainly from the cultivation stage. This is explained by the high uncertain parameters that affect cultivation, namely in terms of N_2O emissions from soil, and fuel and fertilizer inputs;
- uncertainty ranges in cultivation GHG emissions are skewed because N_2O released from cultivated soils is an important contributor to the GHG emissions, and a skewed distribution has been selected to N_2O emissions;
- uncertainty in industrial conversion processes is small, both in energy and GHG terms;
- in industrial conversion processes, energy is a proxy for GHG emissions. This conclusion is not valid at the agricultural stage, due to LUC and N_2O emissions;
- transportation activities hardly contribute to the overall balance of energy and GHG emissions. consecutively

[1] Figures are shown consecutively at the end of this section to facilitate interpretation and comparison.

An in-depth analysis of energy results by life-cycle stage shows that rapeseed oil requires more energy in cultivation than in oil extraction, whereas RME and wheat-based bioethanol require similar amounts for the cultivation and industrial conversion steps. Sugar beet-based bioethanol has the lowest energy consumption in cultivation (per MJ of biofuel produced), because this chain has the highest energy output per hectare. On the other hand, bioethanol from sugar beet shows a particularly high energy requirement at the industrial conversion stage, because the energy required to dry beet pulps has been included in the bioethanol production stage in Fig. 6.6. This is a significant energy consuming step, according to JEC (2008). If beet pulp drying was not included, the energy requirement of the bioethanol production step in Fig. 6.6 would decrease on average from 0.59 to 0.36 MJ MJ^{-1}.

Concerning bioETBE chains, Figs. 6.5, 6.7 and 6.9 show that bioETBE synthesis is particularly energy intensive. This process requires the feedstock isobutylene, which is a by-product of the petroleum refining process. Being of fossil origin, isobutylene has an energy requirement greater than 1. An additional step – bioETBE combustion – is included in Figs. 6.5, 6.7 and 6.9 to take into account the non-renewable share of bioETBE and its contribution to net GHG emissions, as discussed in chapter 4. BioETBE combustion shows particularly high GHG emissions if compared with the remaining steps of the bioETBE life-cycles.

Finally, Figs. 6.8 and 6.9 show E_{req} and GHG intensity results when the sugar pathway is followed in the sugar beet chain (pathway “b” in Fig. 4.5). Main differences to Figs. 6.6 and 6.7 are the higher values in the cultivation and bioethanol production stages, which can be explained by the fact that results are non-allocated and are expressed per MJ of biofuel produced. When sugar production is envisaged, the bioethanol and bioETBE outputs are relatively small, increasing the (per MJ) results: on average 1.25 t of bioethanol (2.6 t of bioETBE) are produced per cultivated hectare, as opposed to ca. 6.8 t bioethanol ha^{-1} (14 t bioETBE ha^{-1}) when sugar beet is fully dedicated to bioethanol and bioETBE production.

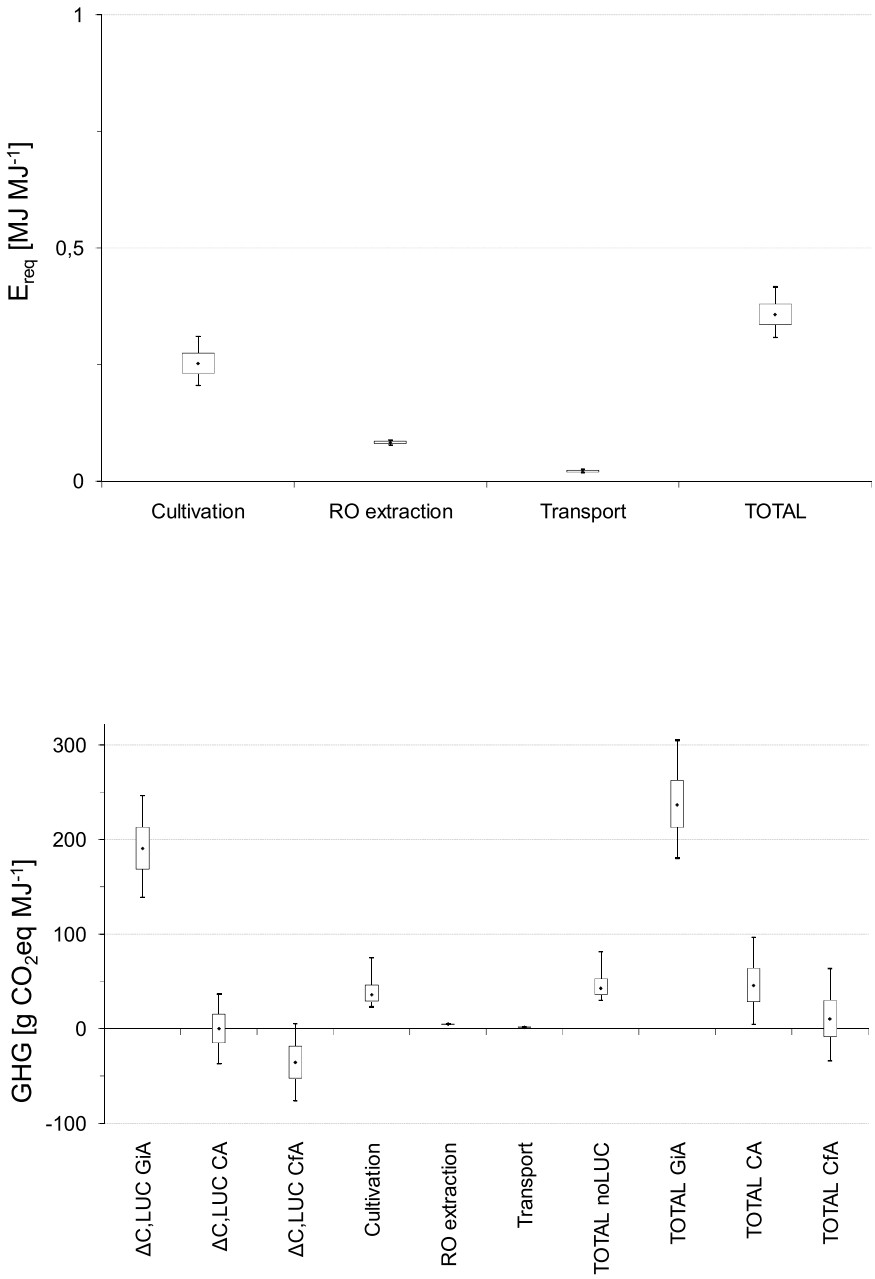


Fig. 6.2. Rapeseed oil: energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values).

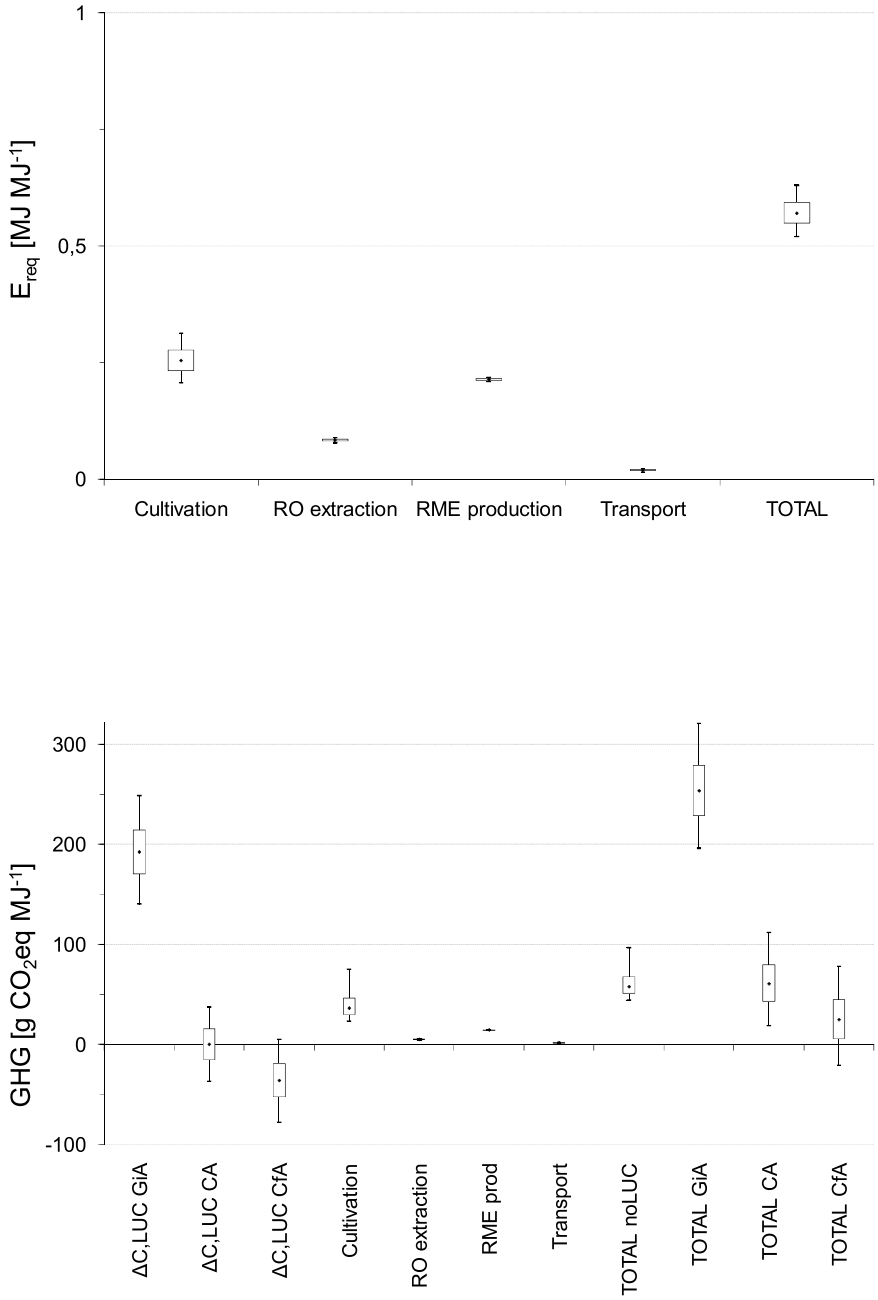


Fig. 6.3. Rapeseed Methyl Ester: energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values).

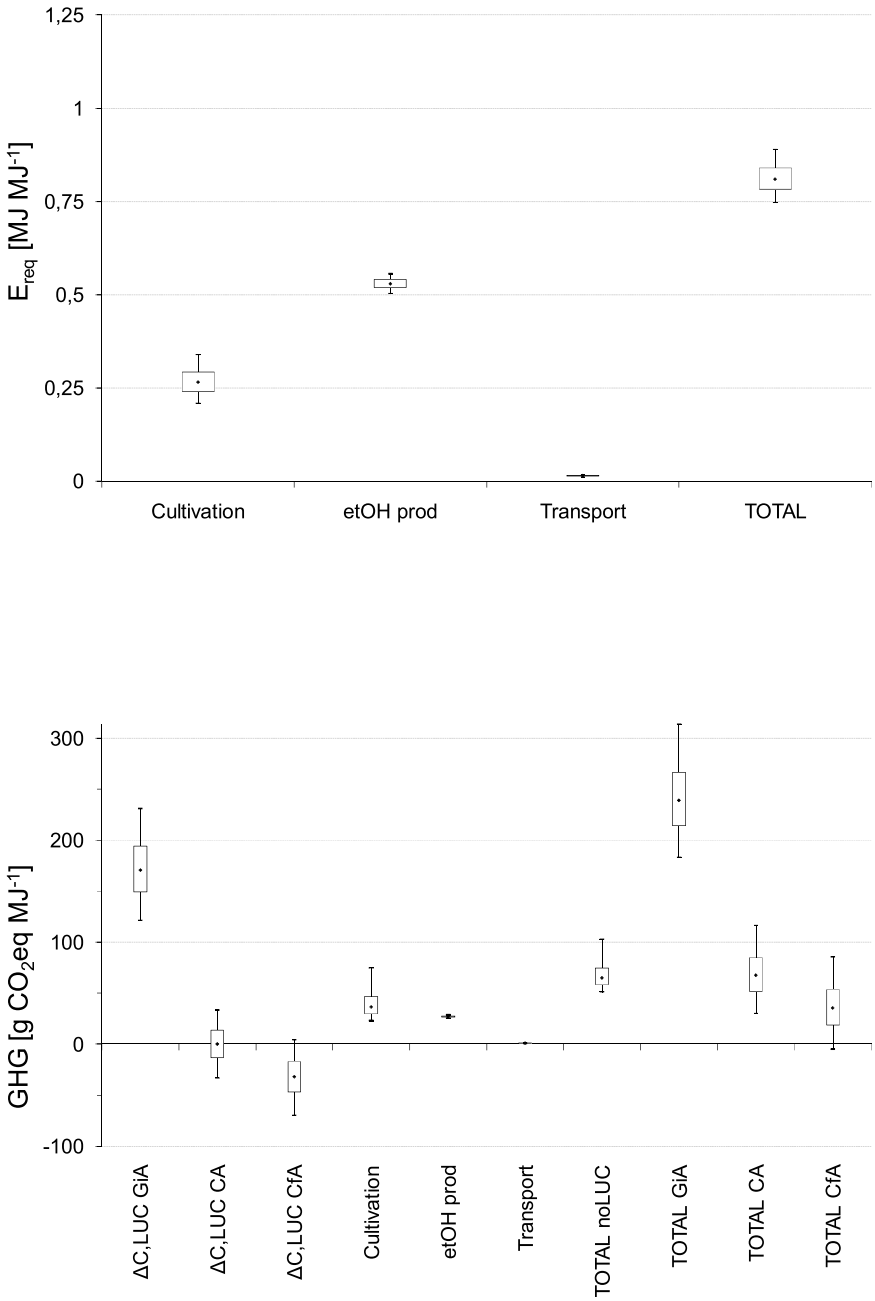


Fig. 6.4. Wheat-based bioethanol: energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values).

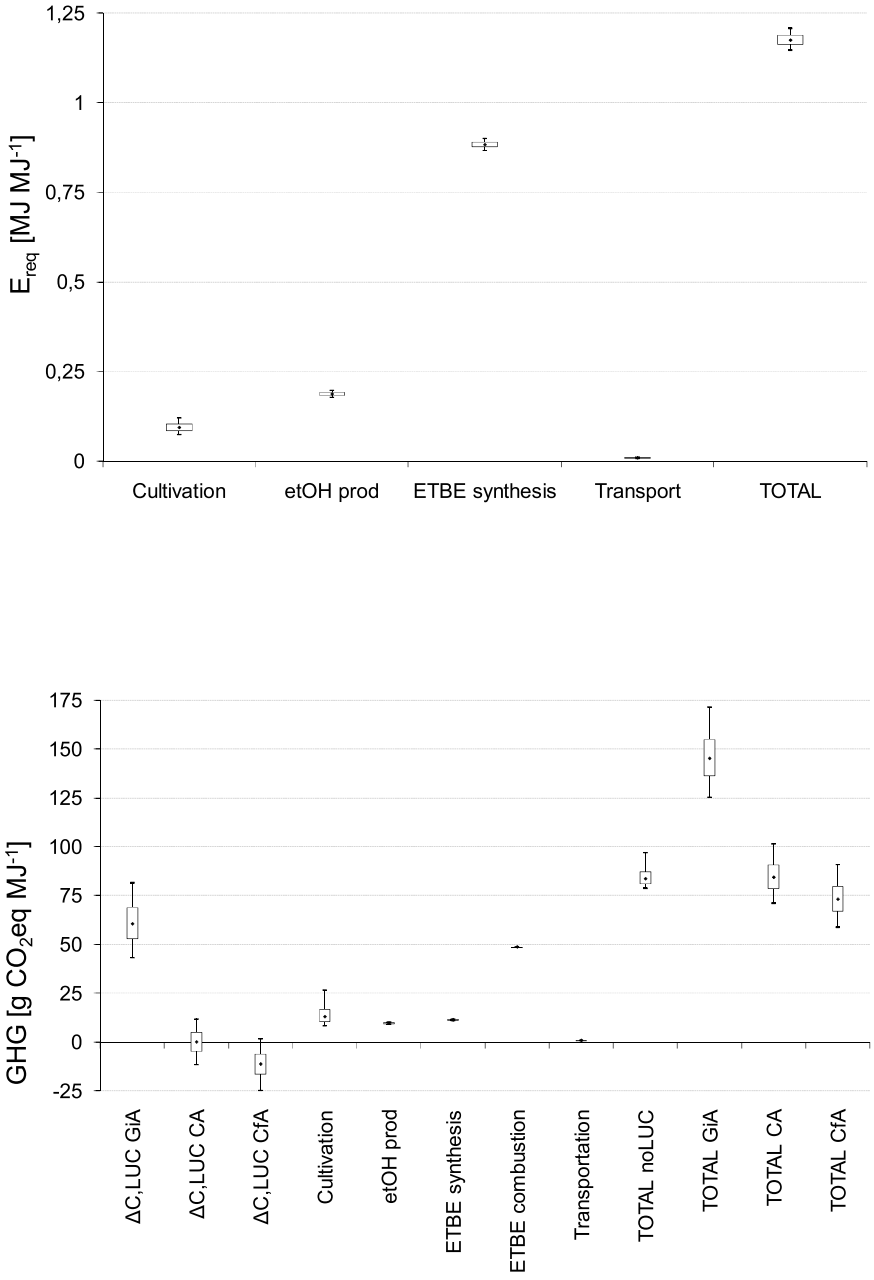


Fig. 6.5. Wheat-based bioETBE: energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values).

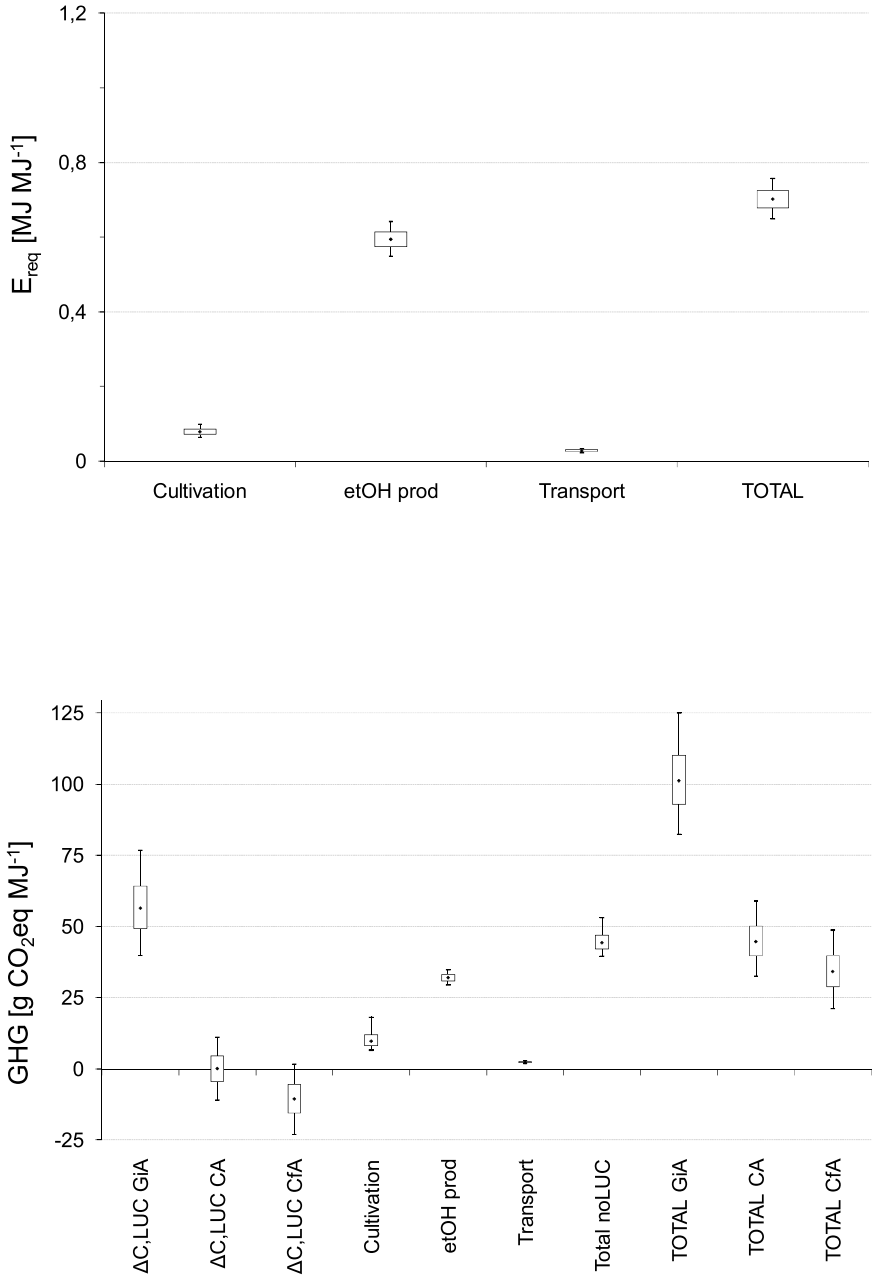


Fig. 6.6. Sugar beet-based bioethanol (bioethanol pathway): energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values). Bioethanol production includes beet pulp drying.

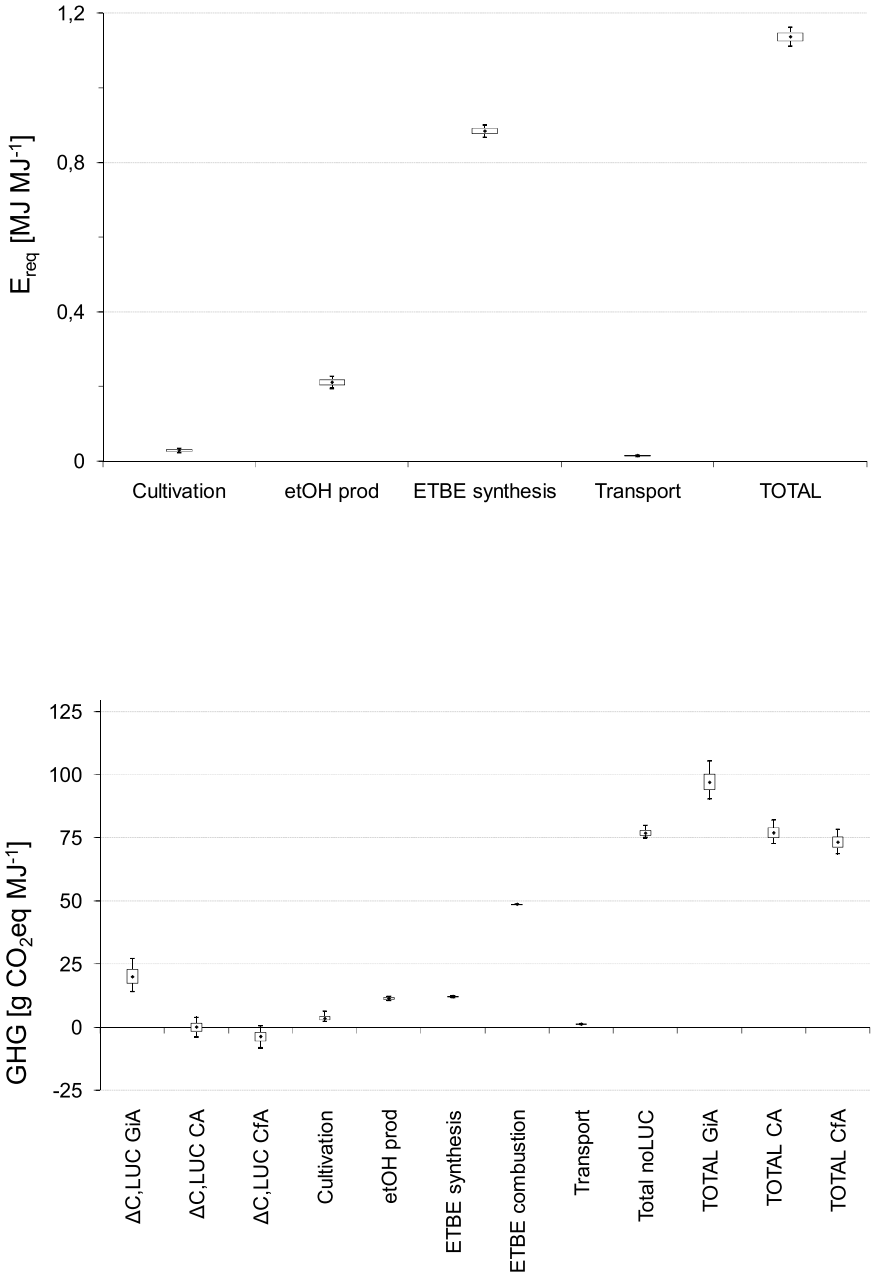


Fig. 6.7. Sugar beet-based bioETBE (bioethanol pathway): energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values).

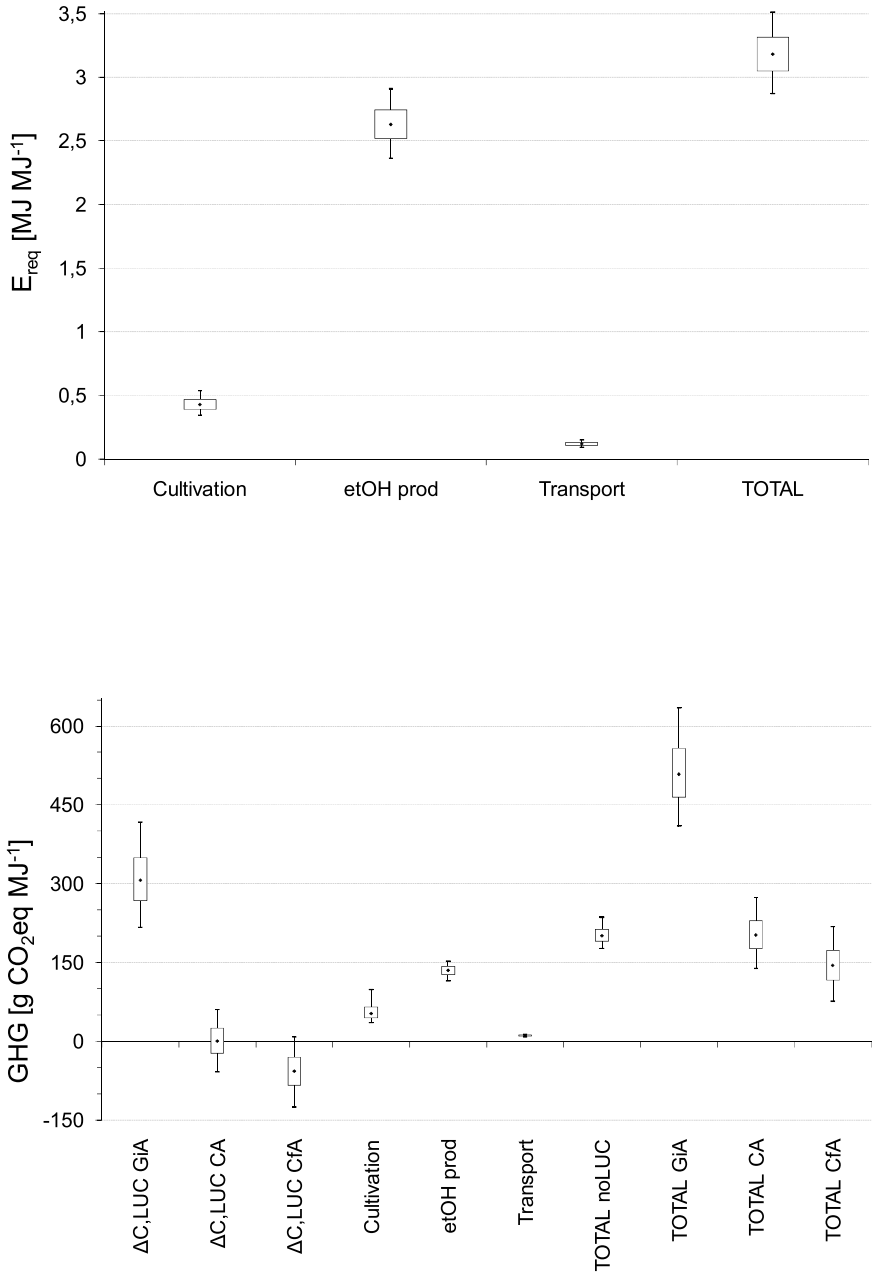


Fig. 6.8. Sugar beet-based bioethanol (sugar pathway): energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values).

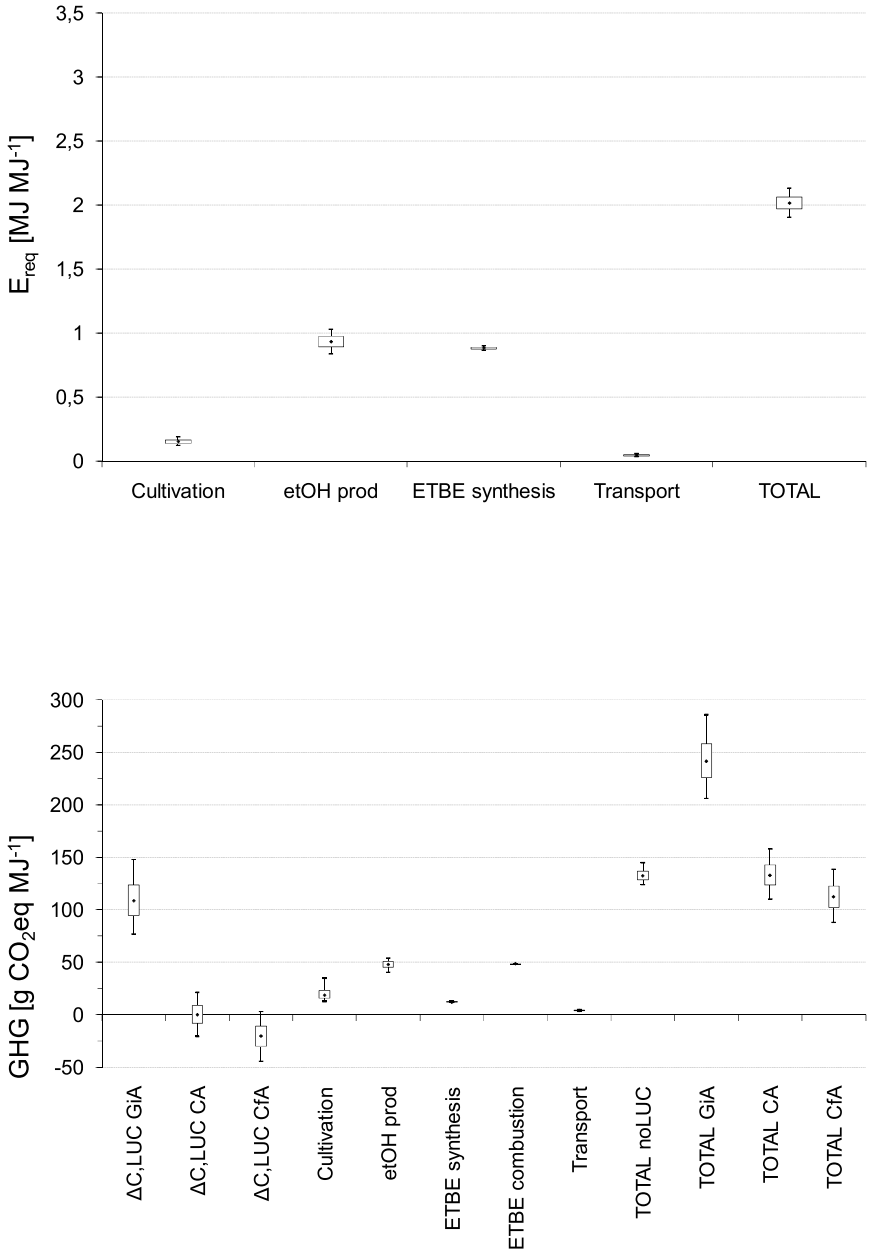


Fig. 6.9. Sugar beet-based bioETBE (sugar pathway): energy requirement (E_{req}) and GHG emissions per life-cycle stage (non-allocated values).

6.3. ENERGY RENEWABILITY EFFICIENCY AND GHG INTENSITY

6.3.1. Introduction

Life-cycle energy renewability and GHG emission results for the biofuel systems addressed in this dissertation are shown in the following sections. Results are displayed in box plots and 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 using mass, energy and market value allocation approaches and the substitution method.

6.3.2. Rapeseed Oil and Rapeseed Methyl Ester

The life-cycle energy renewability efficiency ERenEf of rapeseed oil is displayed in the box plot of Fig. 6.10. 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 58% to 88%, depending on the approach for dealing with co-products) comes from renewable energy sources.

Comparing the three allocation methods used, Fig. 6.10 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.

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.

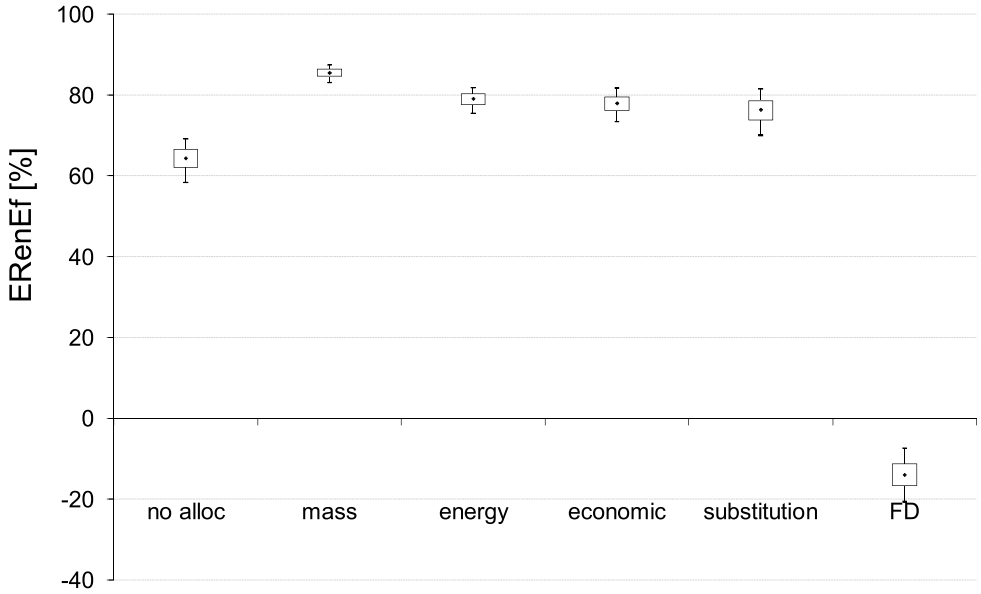
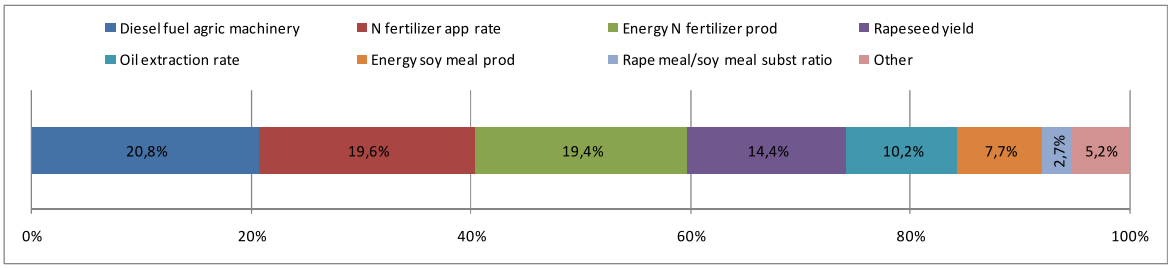
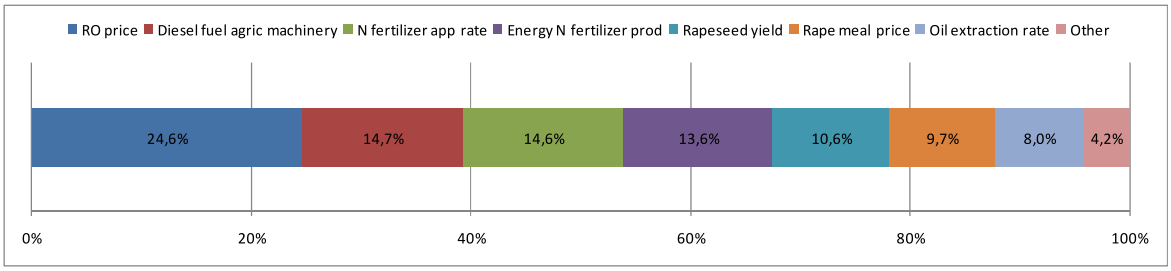


Fig. 6.10. Energy renewability efficiency (ERenEf) of Rapeseed Oil, including reference fossil fuel (fossil diesel FD).

Figure 6.11 shows which parameters are most significant in the overall uncertainty of RO ERenEf using a substitution method (Fig. 6.11a) and economic allocation (Fig. 6.11b). 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. Using the substitution method, the energy associated with producing soy meal and the replacement ratio between soy meal and rape meal also contribute to the variance of results. With economic allocation, Fig. 6.11b) shows that market prices (and their inherent volatility) also affect the variance of RO ERenEf.



(a)



(b)

Fig. 6.11. Contribution of input data to the variance of RO life-cycle ERenEf: (a) substitution method; (b) economic allocation.

The life-cycle GHG intensity of rapeseed oil is displayed in the box plot of Fig. 6.12. An important conclusion from Fig. 6.12 is that parameter uncertainty is significantly higher in the case of RO GHG emissions when compared to ERenEf values of Fig. 6.10. An uncertainty importance analysis will put into evidence the parameters that most contribute to this higher magnitude of uncertainty.

Fig. 6.12 shows that the effect of allocation between co-products is increasingly important as the level of GHG emissions increases. On the other hand, the GHG intensity of rapeseed oil is not sensitive to the substitution method, because the GHG credits associated with soy meal substitution (the product that rape meal displaces) are very low. Soy beans imported from abroad are crushed in the EU, yielding soy meal and soy oil as a co-product (JEC 2007). Soy oil replaces rapeseed oil and thus attracts energy and GHG emission credits to the soy bean chain, which explains the low magnitude of soy meal credits included in the RO chain.

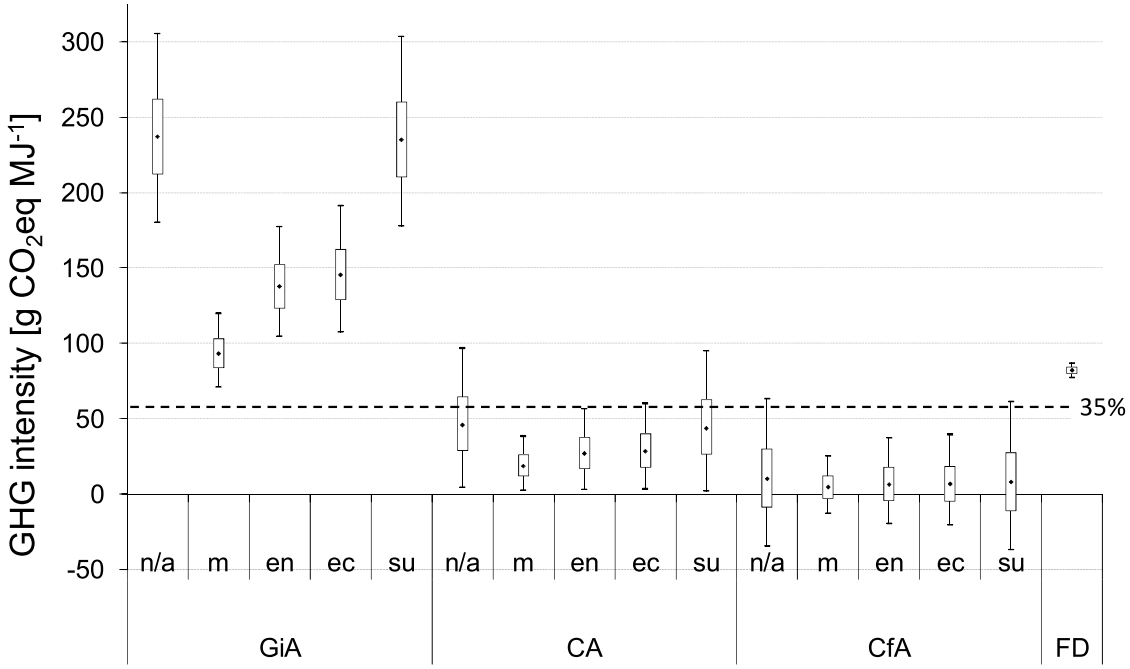
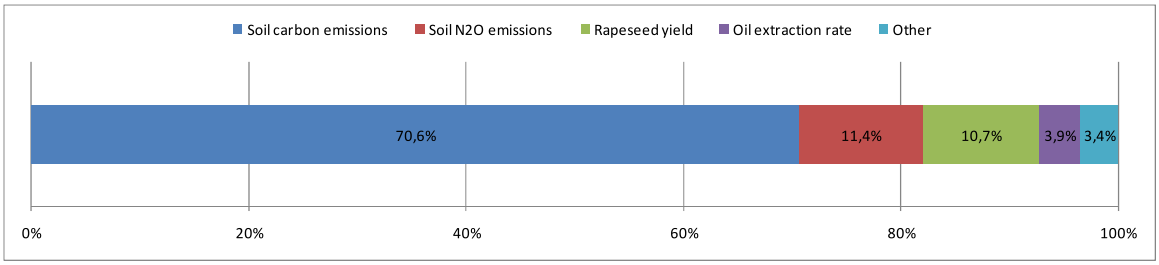


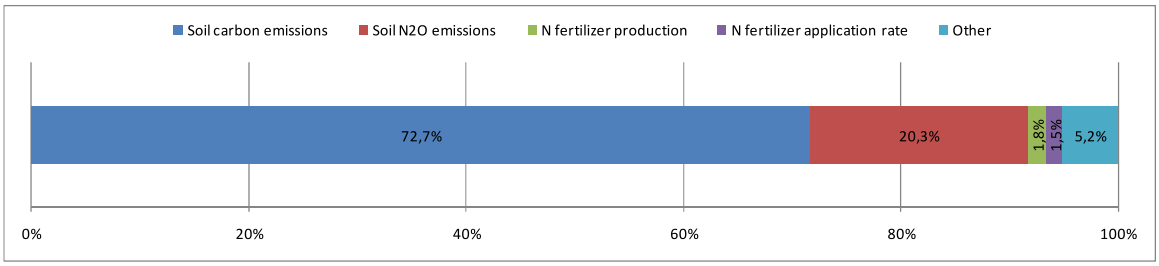
Fig. 6.12. Rapeseed oil life-cycle GHG intensity including fossil diesel (FD). n/a: no allocation; m: mass; en: energy; ec: economic; su: substitution. The dashed line indicates the 35% minimum level of biofuel GHG savings for the purposes of EPC (2009). The same notation is used in similar figures in the remainder of the chapter.

A comparison between rapeseed oil and fossil diesel shows that RO GHG intensity values are considerably higher than FD GHG emissions if the most severe land use change scenario (improved grassland to rapeseed cultivation) is considered. This outcome contrasts with the other two LUC scenarios assessed (conversion from full-tillage or low-tillage croplands). In these two scenarios, rapeseed oil GHG emissions are below FD GHG emissions and comply with the 35% GHG savings of the European renewable energy directive (EPC 2009), regardless of the co-product method used. Fig. 6.12 also shows that in the "low-tillage cropland to rapeseed cultivation" LUC scenario (CA), the 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" (CA scenario) results in very low RO GHG emissions. For this reason, differences between co-product approaches become negligible.

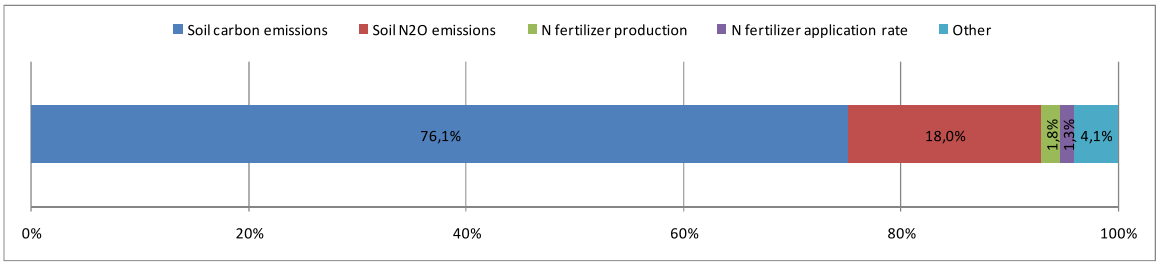
Figure 6.13 shows which parameters are most significant in the overall uncertainty of RO GHG emissions for the three LUC scenarios considered. The highest sources of uncertainty arise in the cultivation stage. Soil carbon emissions from land use change are the main contributor to the uncertainty of RO 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 GiA LUC scenario. The remaining parameters hardly contribute to the variance of GHG emissions.



(a)



(b)



(c)

Fig. 6.13. Contribution of input data to the variance of RO life-cycle GHG intensity (substitution method). Land use change scenarios: (a) GiA; (b) CA; (c) CfA.

Figures 6.14 and 6.15 show the energy renewability efficiency and GHG intensity of Rapeseed Methyl Ester (RME). In comparison to the RO production system, RME production requires the additional step of transesterification, which motivates that RME

ERenEf and GHG results are, respectively, lower and higher than RO results. Comparing Figs. 6.10, 6.12 (RO) with Figs. 6.14, 6.15 (RME), it can be seen that the same trends are displayed in terms of scenario and parameter uncertainty. Moreover, an uncertainty importance analysis has been conducted, showing that uncertainty ranges are due to the same driving factors, particularly soil emissions at the cultivation stage. Application of the substitution method has little effect in the GHG intensity of RME, because the GHG credits associated with rape meal displacing soy meal and the use of glycerin as animal feed have both relatively low values.

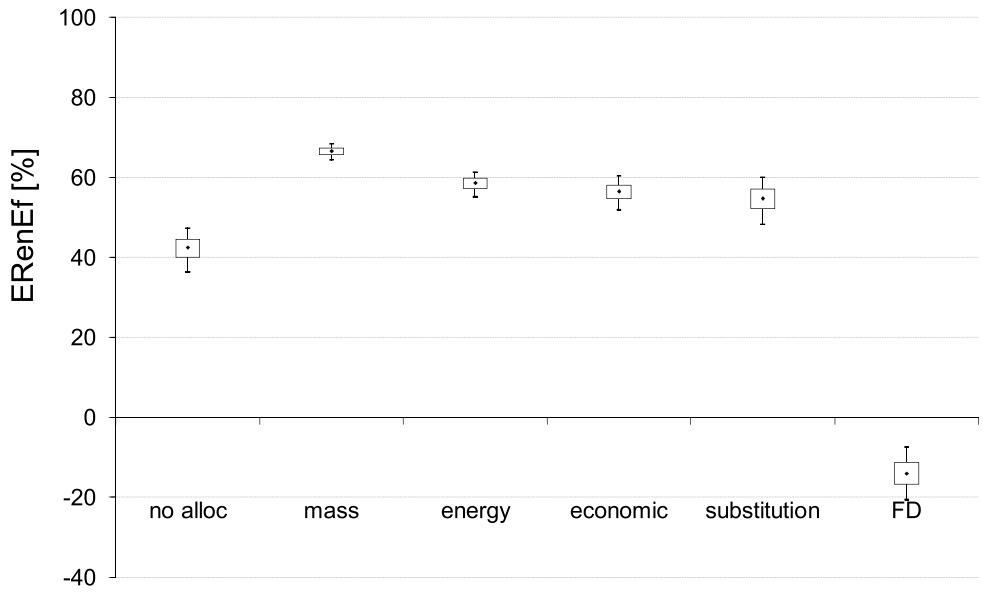


Fig. 6.14. Energy renewability efficiency (ERenEf) of RME, including reference fossil fuel (fossil diesel FD). Substitution method: glycerin as animal feed.

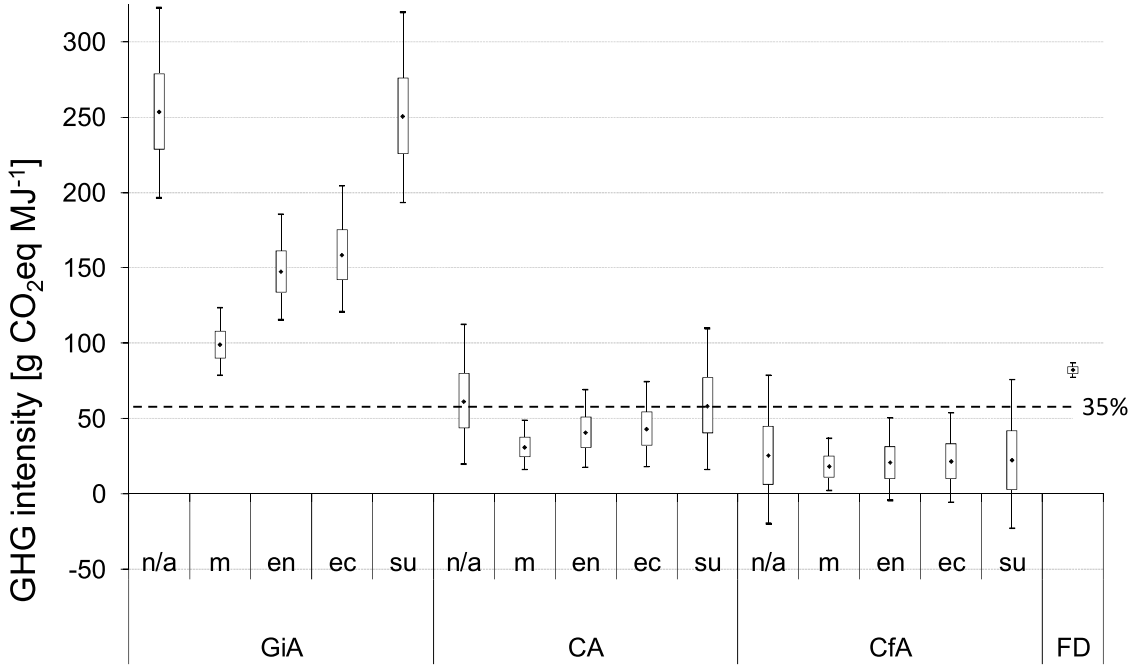


Fig. 6.15. RME life-cycle GHG intensity including fossil diesel (FD). Substitution method (su): glycerin as animal feed.

An additional difference between RO and RME arises from the alternative uses that the transesterification co-product glycerin may have (Figs. 6.16 and 6.17). No significant differences are observed between three potential uses for glycerin –as animal feed, for process heat, or displacing propylene glycol–, because the energy and GHG credits are quite similar for these options. The ranges of GHG emissions calculated for each scenario clearly overcome the differences between scenarios. Therefore, it is difficult to definitely point out a most favorable option on the basis of a GHG emissions criterion.

Concerning the use of co-product glycerin as a replacement for synthetic glycerin, ERenEf results are significantly higher. An average energy renewability efficiency of 98% is calculated for RME when credits for synthetic glycerin substitution are accounted for, indicating that RME production is nearly completely renewable in energy terms. This outcome is due to the highly energy intensive production of synthetic glycerin, which requires about 18 times its heating value in fossil fuel. In particular, the 75th and 95th percentiles are in this case higher than 100%, which indicates that the energy inputs for

RME production are more than offset by the energy required for synthetic glycerin production, i.e. the net energy consumption of the RME life-cycle actually becomes negative.

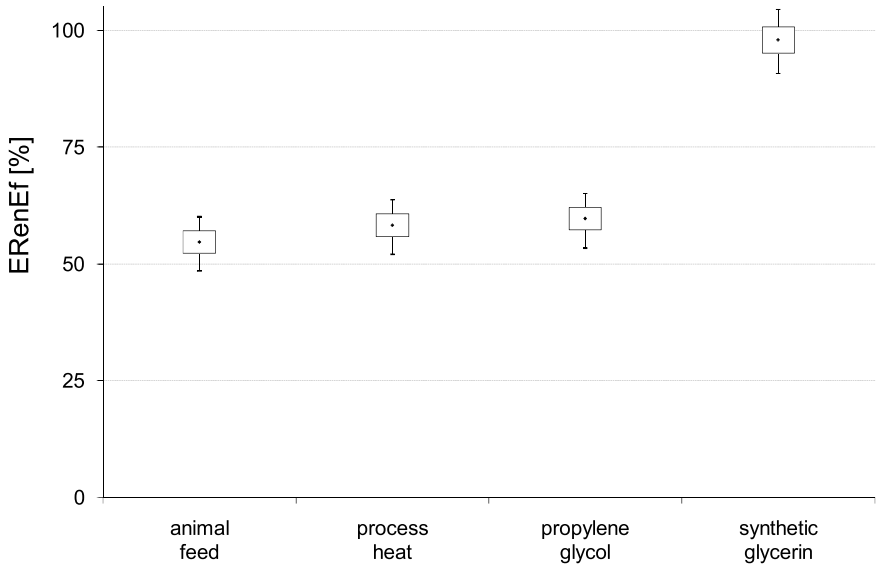


Fig. 6.16. Energy renewability efficiency of RME for alternative co-product glycerin uses.

GHG intensity of RME for different glycerin uses are similar to ERenEf results, with the synthetic glycerin option deviating from the other three alternatives. In particular, negative GHG emissions (5^{th} percentile in the right column of Fig. 6.17) are calculated when synthetic glycerin credits are considered in the substitution method. This result is consistent with findings from e.g. Rollefson et al. (2004), which indicate negative values for biodiesel GHG emissions when synthetic glycerin is used for calculation of co-product credits.

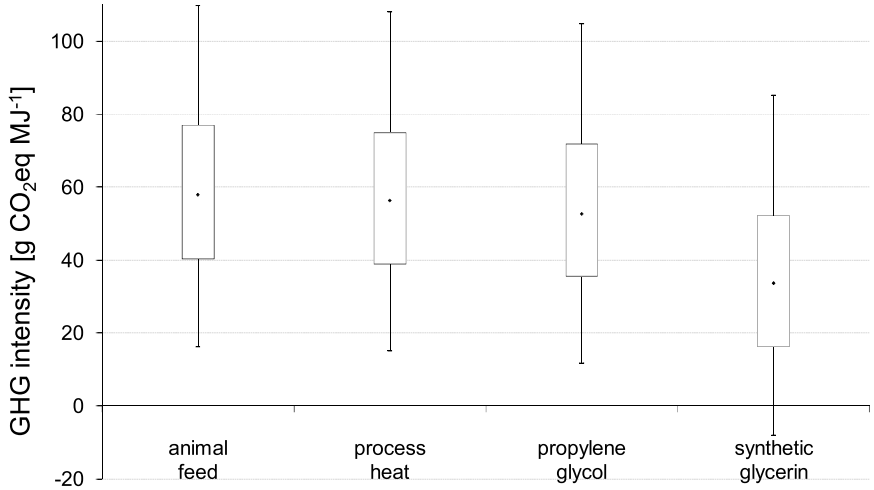


Fig. 6.17. RME life-cycle GHG intensity for alternative co-product glycerin uses. (LUC scenario: CA).

6.3.3. Wheat-based bioethanol and bioETBE

The life-cycle energy renewability and GHG emissions of wheat-based bioethanol are displayed in the box plots of Figs. 6.18 and 6.19. These figures show that parameter uncertainty is significantly higher in the case of GHG emissions when compared to ERenEf figures. In particular, in the "full-tillage cropland to wheat cultivation" LUC scenario (CA, Fig. 6.19), the uncertainty range clearly overcomes the differences between calculated median values for the various scenarios of dealing with co-products. An uncertainty importance analysis will put into evidence the parameters that most contribute to this level of uncertainty.

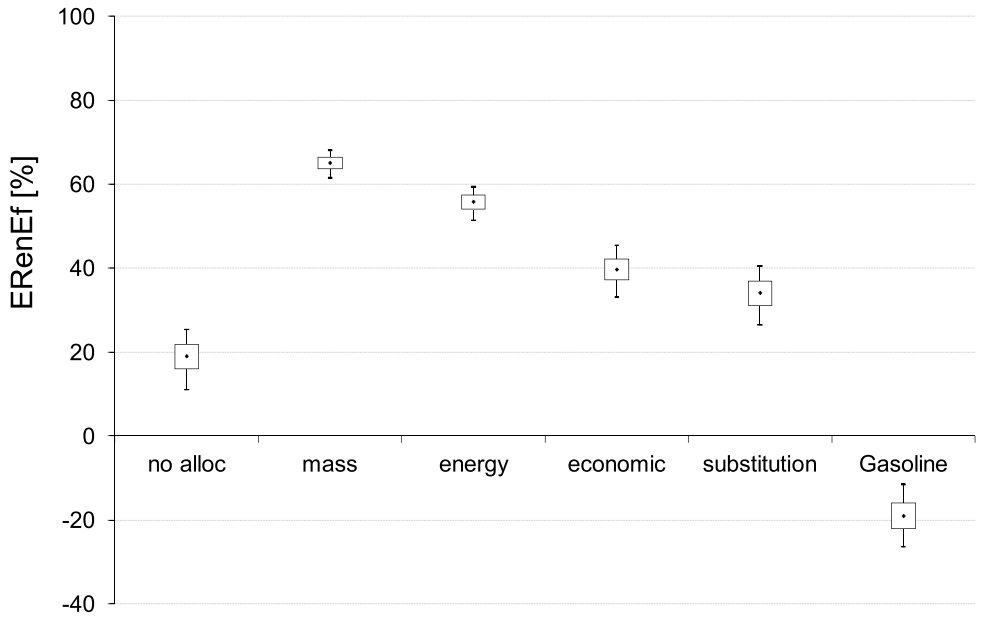


Fig. 6.18. Energy renewability efficiency (ERenEf) of wheat-based bioethanol, including reference fossil fuel (gasoline).

A comparison with the reference fossil fuel that bioethanol mainly displaces –gasoline– shows that bioethanol clearly contributes to non-renewable primary energy savings as opposed to its fossil reference. Bioethanol ERenEf is clearly positive, which indicates that an important fraction of the biofuel energy content (from 50% to 70%, depending on the approach for dealing with co-products, Fig. 6.18) comes from renewable energy sources. This outcome contrasts with the GHG intensity values of Fig. 6.19. Bioethanol may have considerably higher GHG emissions than gasoline, if the most severe land use change scenario (improved grassland converted to wheat cultivation, GiA) is considered. In this case, it can take several years of bioethanol displacing gasoline to compensate for the increased emissions associated with LUC (carbon payback time), as demonstrated in section 6.5.

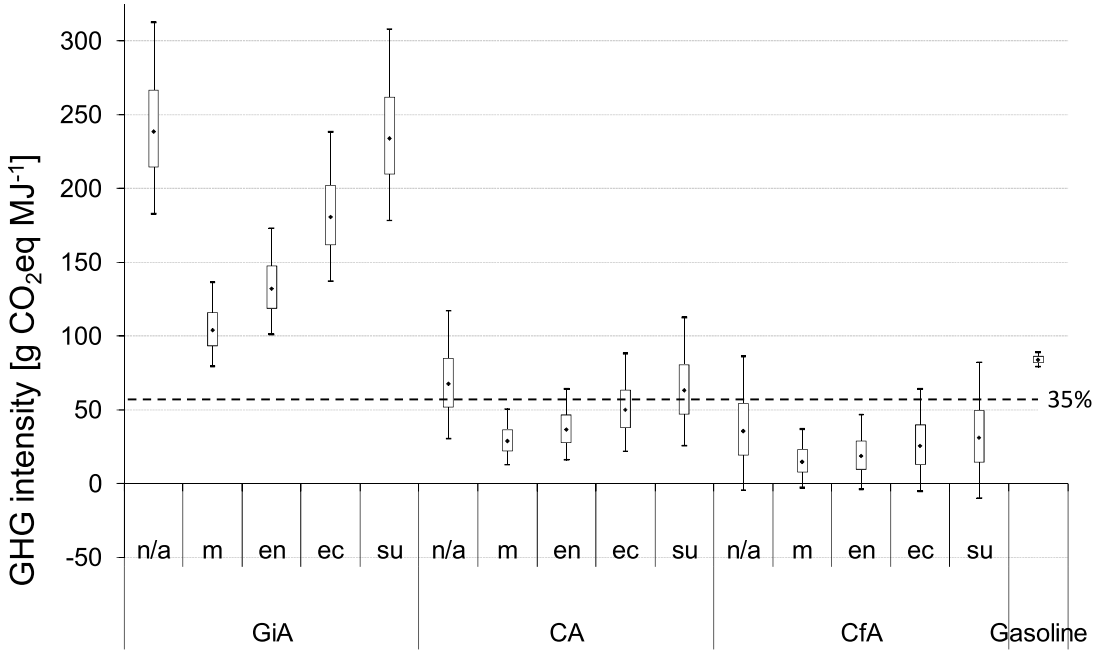


Fig. 6.19. Life-cycle GHG intensity of wheat-based bioethanol, including reference fossil fuel (gasoline).

Figure 6.20 shows which parameters are most significant in the overall uncertainty of ERenEf and GHG emissions of wheat-based bioethanol. Concerning the energy renewability efficiency, the uncertainty importance analysis that has been conducted shows that several (from 4 to 6) parameters have important contributions in the uncertainty, namely wheat yield, N fertilizer application rate, and energy consumption in N fertilizer production and fermentation and distillation stages. In particular, Fig. 6.20(b) for economic allocation shows that market prices (and their inherent volatility) also affect the variance of ERenEf. Concerning the uncertainty in bioethanol GHG emissions (Fig. 6.20c), results show that the highest sources of uncertainty arise in the cultivation stage. Soil carbon emissions from land use change are clearly the main contributor to the uncertainty of bioethanol GHG intensity, with nitrous oxide emissions from cultivated soil as the second most important aspect. The remaining parameters hardly contribute to the variance of GHG emissions. Further research work must focus on the highlighted sources of uncertainty, in order to reduce the overall uncertainty of the bioethanol chain and improve the reliability of life-cycle studies outcomes.

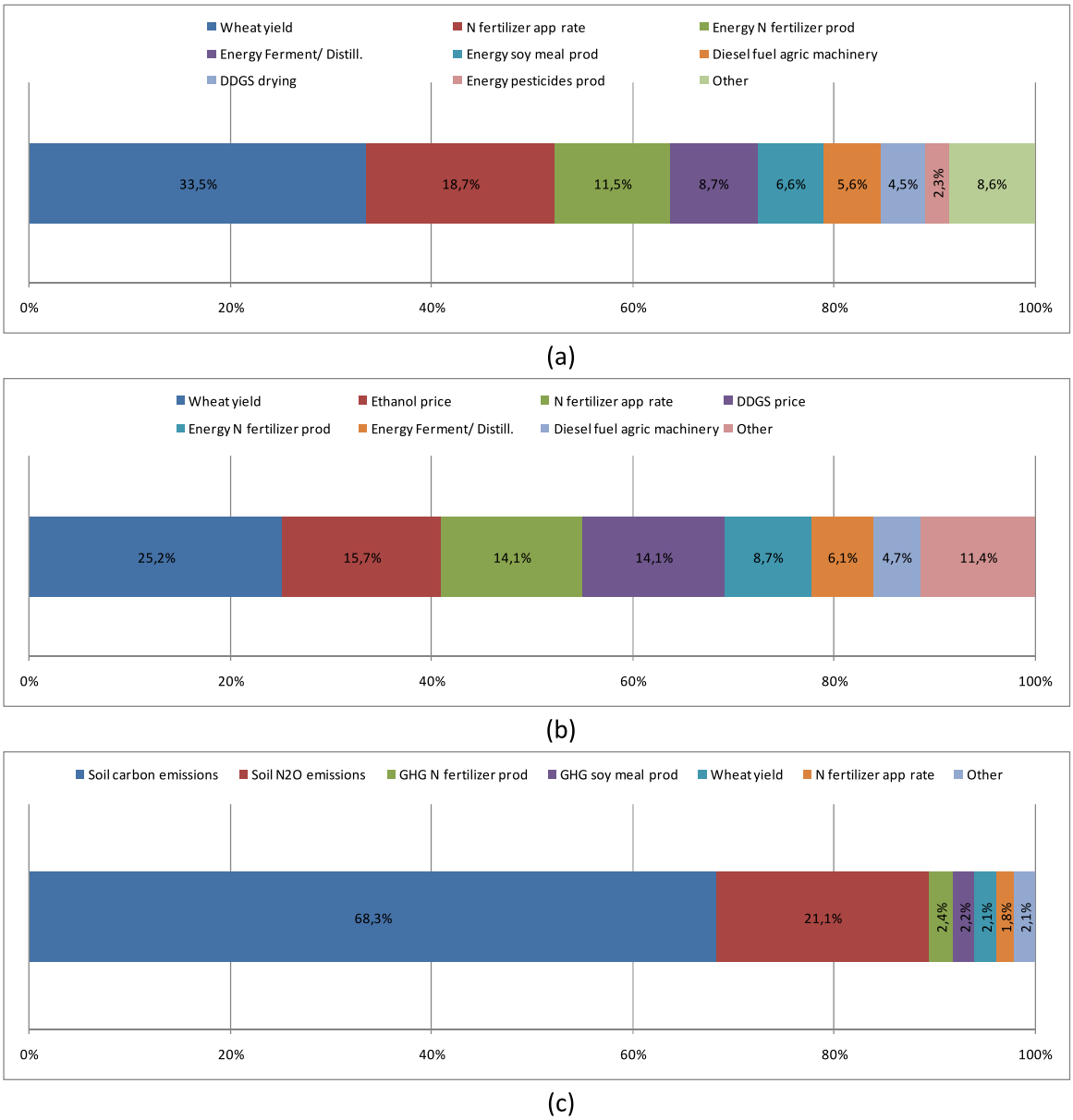


Fig. 6.20. Wheat-based bioethanol: Contribution of input data to the variance of (a) ERenEf – substitution method; (b) ERenEf – economic allocation; and (c) GHG intensity – substitution method (LUC scenario: CA).

The energy renewability efficiency and GHG intensity of wheat-based bioETBE are shown in Figs. 6.21 and 6.22, respectively. Compared with bioethanol results (Figs. 6.18 and 6.19), the ERenEf of bioETBE is significantly lower. With the exception of mass allocation, ERenEf results are negative, which indicates that bioETBE is a non-renewable fuel. This can be explained by the energy-intensive process of bioETBE synthesis, as already shown

in section 6.2. Nonetheless, bioETBE is still advantageous over gasoline, which shows the lowest ERenEf value.

Concerning the GHG intensity of bioETBE, Fig. 6.22 shows that the trends already observed for bioethanol in Fig. 6.19 still remain, namely the differences between LUC scenarios and between co-product approaches within each LUC scenario. However, an additional feature when comparing Figs. 6.22 and 6.19 is the "translational" effect of the additional refinery step of bioethanol conversion into bioETBE: when bioethanol GHG emissions are relatively high (GiA scenario), the additional step of bioETBE synthesis (with its own GHG emissions) has a damping effect in the overall life-cycle, leading to a lower level of emissions for bioETBE; conversely, when the GHG emissions of the bioethanol life-cycle are lower (CA and CfA LUC scenarios), adding the bioETBE synthesis step raises the overall GHG emissions.

Fig. 6.22 shows that only small GHG emission savings can be achieved when bioETBE displaces gasoline, and just for the CA and CfA LUC scenarios. In these two scenarios, the GHG savings achieved by bioethanol production are partly reduced by the fact that a fraction (53% m/m) of bioETBE comes from fossil sources (refinery by-product isobutylene). The dashed line of 35% GHG emission savings is shown for reference purposes only, as Directive 2009/28/EC only requires that this target be achieved by the part from renewable sources of bioETBE (i.e. bioethanol).

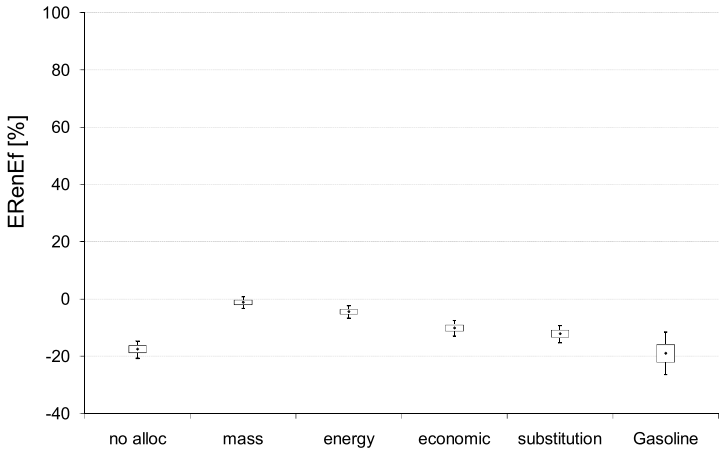


Fig. 6.21. Energy renewability efficiency (ERenEf) of wheat-based bioETBE, including reference fossil fuel (gasoline).

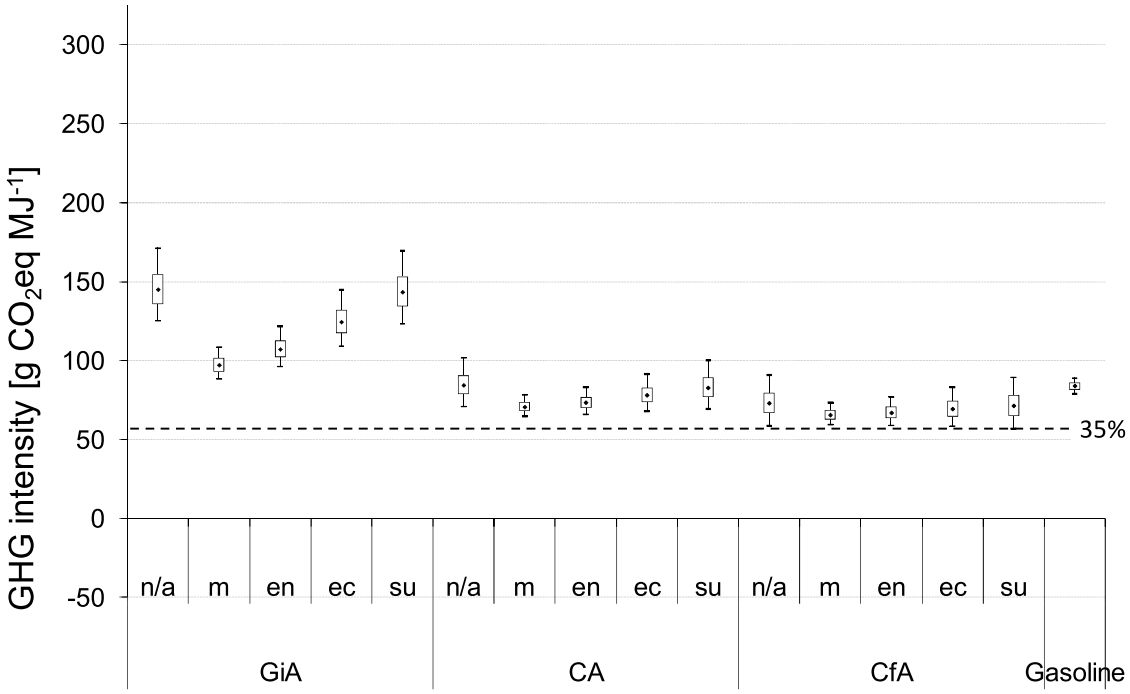


Fig. 6.22. Life-cycle GHG intensity of wheat-based bioETBE, including reference fossil fuel (gasoline).

6.3.4. Sugar beet-based bioethanol and bioETBE

The life-cycle energy renewability efficiency of sugar beet-based bioethanol and bioETBE is shown in Fig. 6.23 for the two pathways addressed in this dissertation (cf. Fig. 4.5): (i) the bioethanol pathway (Figs. 6.23a,b); and (ii) the sugar pathway (Figs. 6.23c,d). Concerning the bioethanol pathway, it can be seen that bioethanol ERenEf results are relatively low in comparison to the ERenEf values of the other biofuel systems addressed in this dissertation, which is partially attributed to the energy input required to dry the sugar beet pulps[2]. Nevertheless, ERenEf values are significantly higher than gasoline ERenEf and do not present significant variations when different co-product approaches are considered. The energy-intensive process of bioETBE synthesis brings the overall energy renewability efficiency to negative values, very close to gasoline´s ERenEf. With

[2] Sugar beet pulps are dried before use as animal feed. The energy input to this process is higher than the credit for the fodder saved. Nevertheless, this is the most likely destination for the co-product on economic grounds (JEC 2007).

regard to uncertainty ranges, these are relatively low for bioethanol (Fig. 6.23a) and become even lower when the additional step of bioETBE production is included (Fig. 6.23b), due to the lower level of uncertainty associated with the isobutylene feedstock.

On the other hand, when sugar beet is processed for sugar production, large variations between co-product approaches are shown. These variations are explained by the high mass share of sugar in this pathway (sugar is the intended product) and the disparities in the allocation data between sugar and bioethanol. In particular, results with economic allocation in the sugar pathway are lower than results with the other allocation approaches due to the lower price of sugar with respect to bioethanol. Concerning the substitution method, Fig. 6.23c shows that the substitution credits for sugar, as described in chapter 4, introduce an additional degree of uncertainty in the results. It is interesting to note that due to the large differences between co-product approaches in the sugar pathway, the inclusion of the bioETBE synthesis process lowers ERenEf results in the case of allocation, but increases ERenEf values in the case of substitution. The conversion of bioethanol into bioETBE also tightens the uncertainty ranges, as explained before.

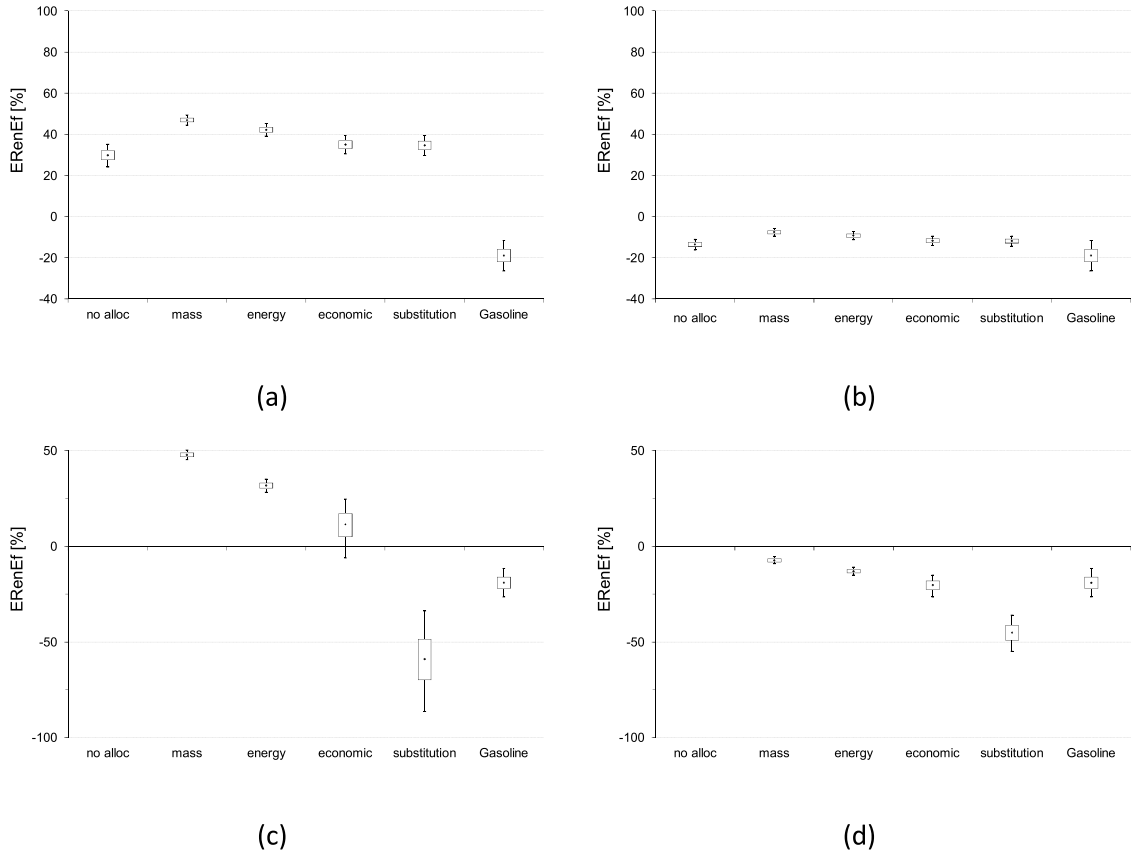
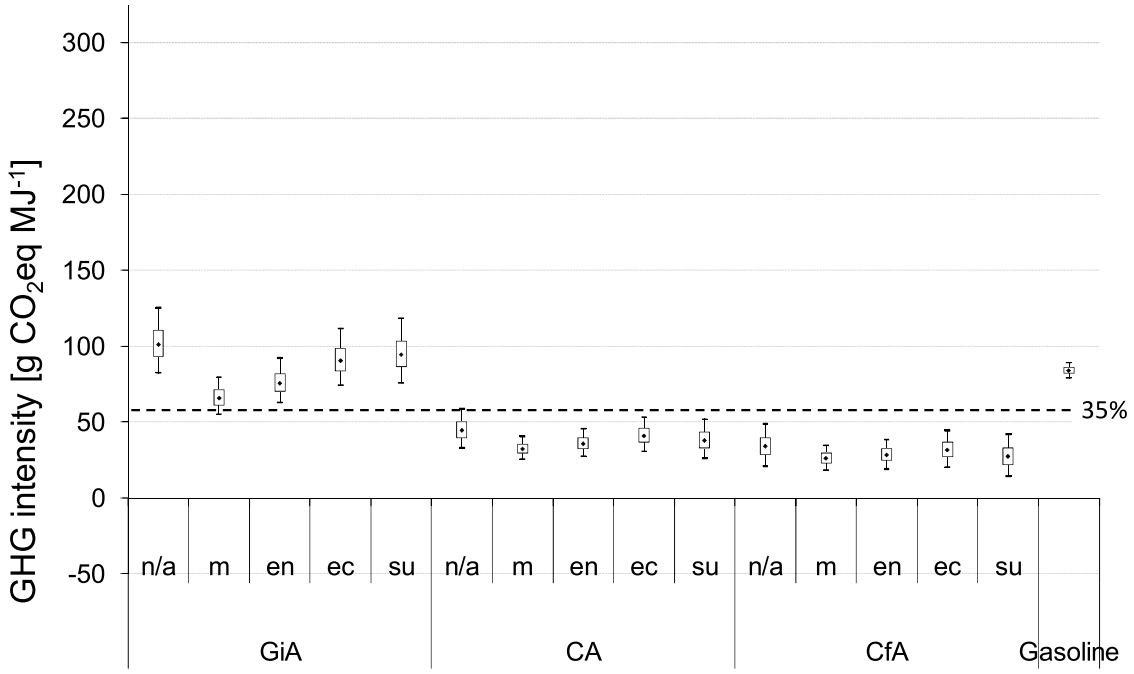
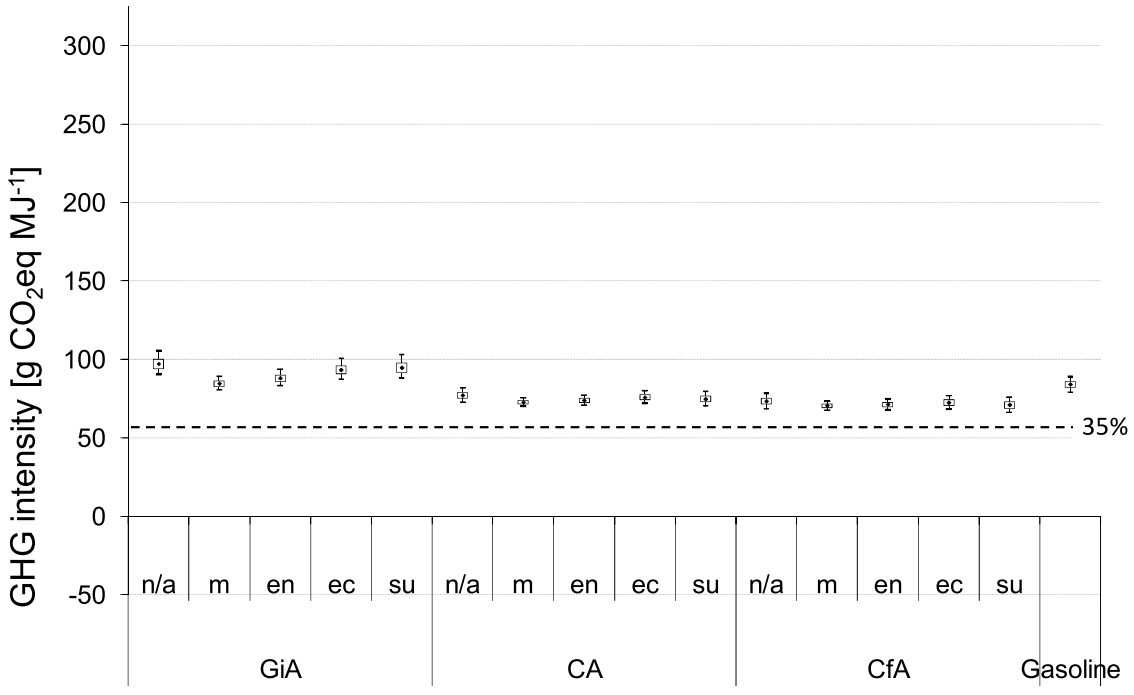


Fig. 6.23. Energy renewability efficiency (ERenEf) of sugar beet-based: (a) bioethanol; (b) bioETBE; (c) bioethanol (sugar pathway); and (d) bioETBE (sugar pathway). Non-allocated values in the sugar pathway ($median_{bioethanol}$ = -218%; $median_{bioETBE}$ = -101%) are not shown to improve the visibility and comparability of the remaining points.

The GHG intensity of sugar beet-based bioethanol and bioETBE is shown in Figs. 6.24 and 6.25, for the bioethanol and sugar pathways, respectively. Fig. 6.24 shows the trend already presented for ERenEf values: moving from bioethanol to bioETBE production reduces the uncertainty range of GHG emissions, and may increase or decrease calculated median values depending on the magnitude of bioethanol life-cycle emissions. Concerning the sugar pathway (Fig. 6.25), large variations are shown between co-product approaches, as already discussed for ERenEf results. Moreover, GHG credits introduced by the substitution method may lead to very low (or even negative) bioethanol life-cycle GHG emissions in the most favorable LUC scenarios (CA and CfA).



(a)



(b)

Fig. 6.24. Life-cycle GHG intensity of sugar beet-based: (a) bioethanol; and (b) bioETBE (bioethanol pathway).

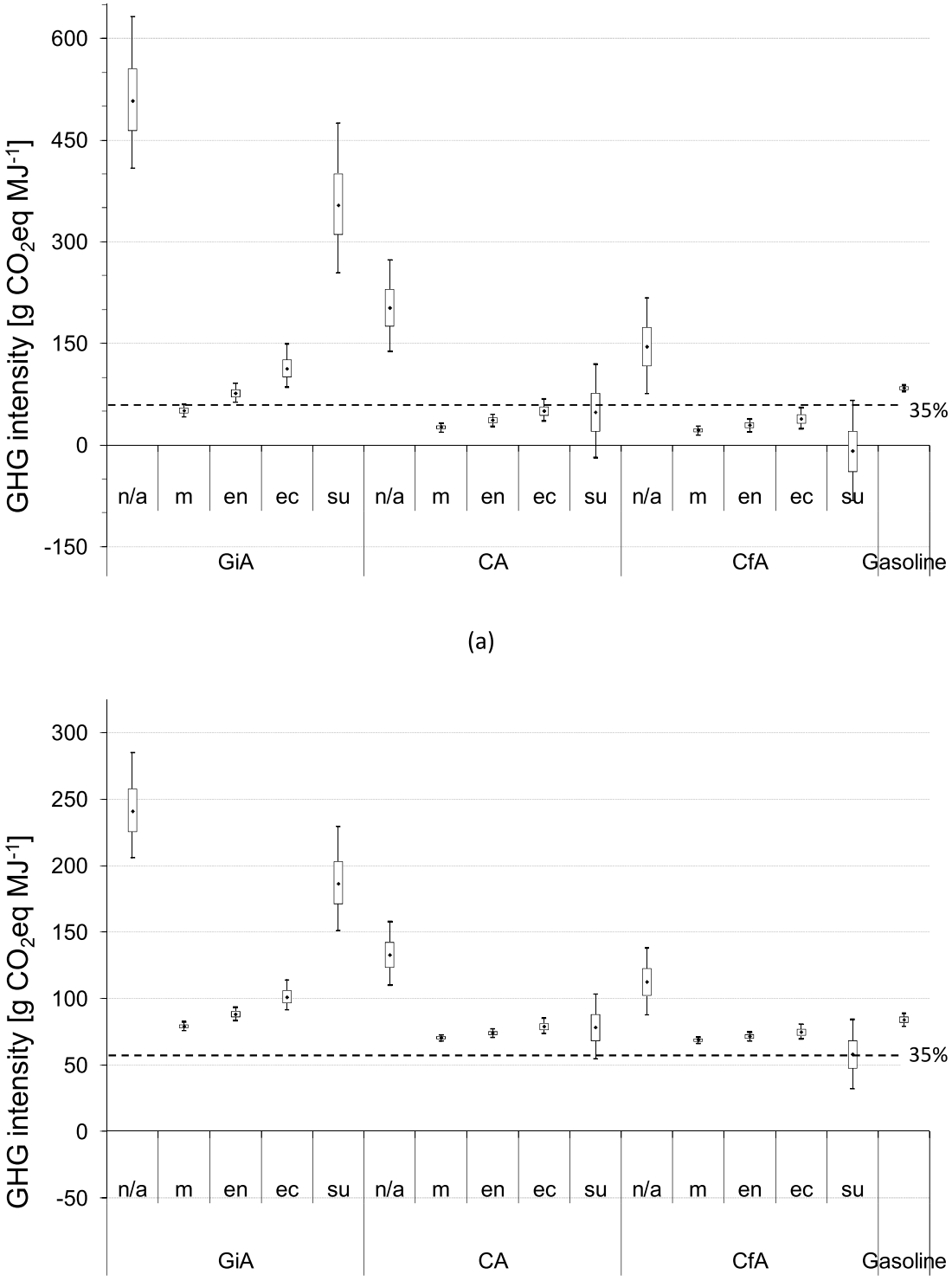
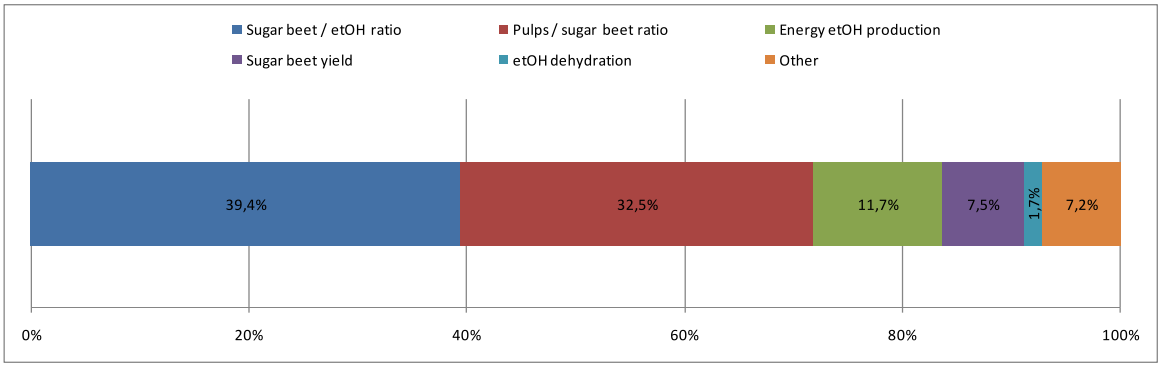


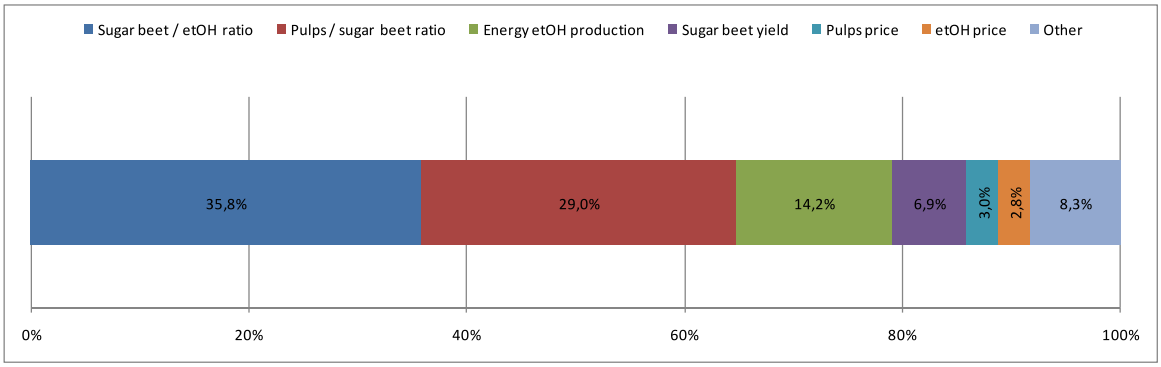
Fig. 6.25. Life-cycle GHG intensity of sugar beet-based (sugar pathway): (a) bioethanol; and (b) bioETBE. Different scales at the yy-axis may difficult direct comparison of uncertainty ranges.

Results of the contribution importance analysis conducted for sugar beet-based bioethanol are shown in Figs. 6.26 and 6.27, respectively for the bioethanol and sugar

pathways. The mass ratios of the chain are major contributors to the variance of the energy renewability efficiency ERenEf, except for the sugar pathway (economic allocation) in which sugar price is the most important parameter. Concerning GHG intensity, carbon and nitrous oxide emissions from soils are the most important factors affecting the variance in the bioethanol pathway, whereas in the sugar pathway soil carbon emissions and GHG credits of imported sugar are the most significant.



(a)



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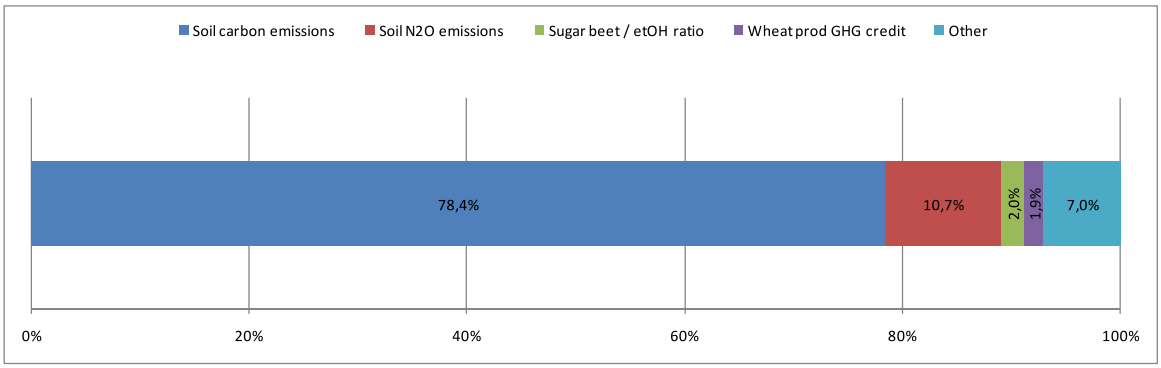
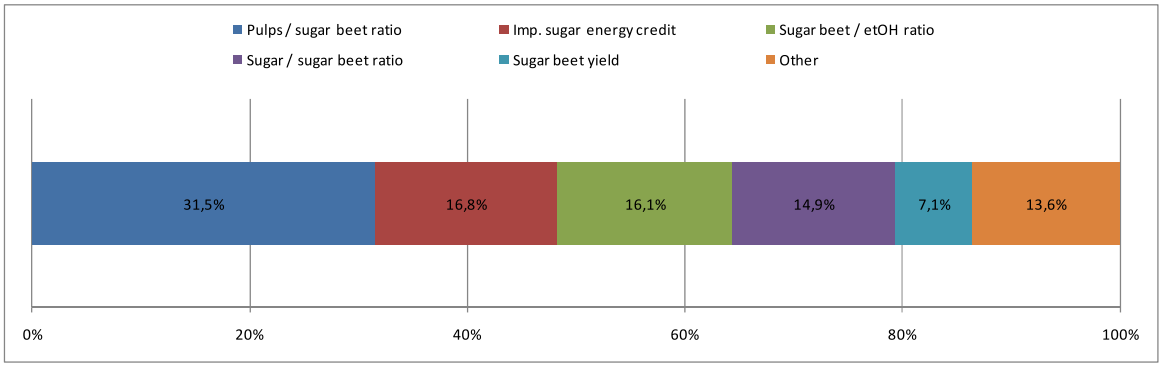
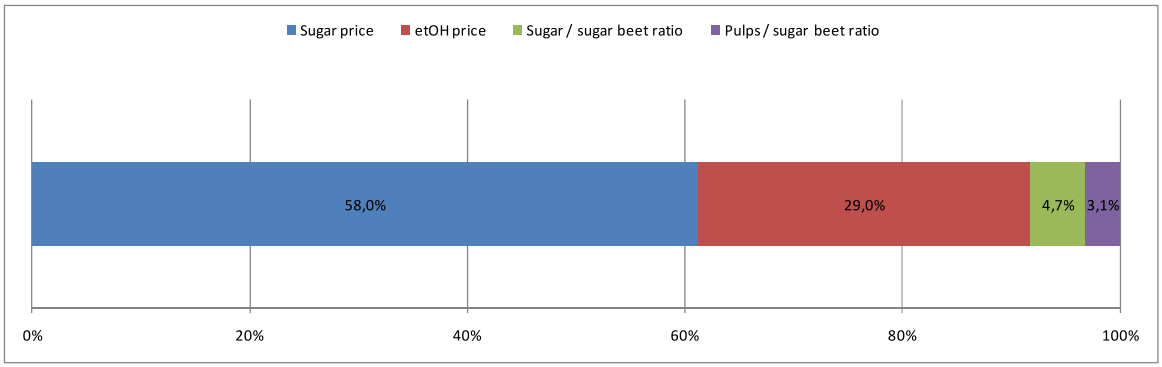


(c)

Fig. 6.26. Sugar beet-based bioethanol (bioethanol pathway): Contribution of input data to the variance of (a) ERenEf – substitution method; (b) ERenEf – economic allocation; and (c) GHG intensity – substitution method (LUC scenario: CA).



(a)



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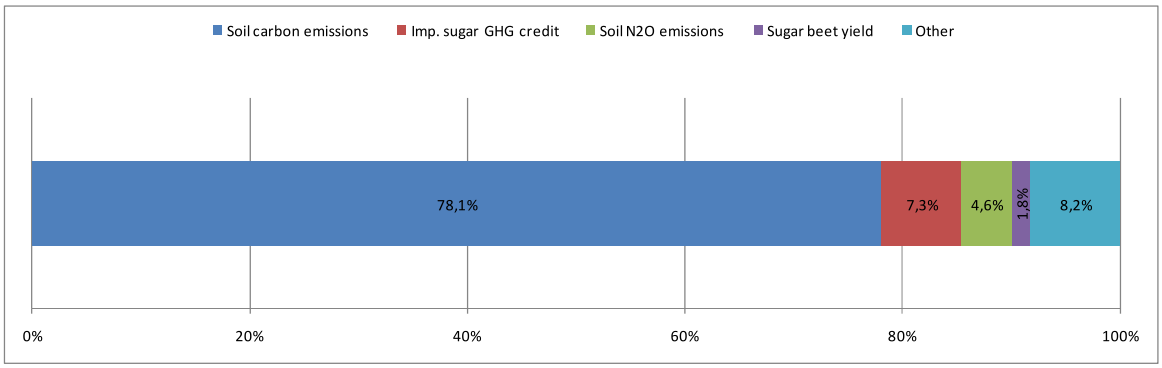


(c)

Fig. 6.27. Sugar beet-based bioethanol (sugar pathway): Contribution of input data to the variance of (a) ERenEf – substitution method; (b) ERenEf – economic allocation; and (c) GHG intensity – substitution method (LUC scenario: CA).

Results of the contribution importance analysis conducted for bioETBE are not shown, due to similarity with bioethanol results. The exception is for ERenEf in the bioethanol pathway, in which the energy consumption of the bioETBE synthesis process also

contributes to the variance with, respectively, 4.4% (substitution method) and 5.4% (economic allocation).

6.3.5. Comparison between biofuel production systems

Comparing the life-cycle energy renewability efficiency and GHG intensity results presented in the last sections, several important conclusions can be drawn. Aspects that are common to the biofuel systems under analysis include:

- allocation approaches for treating co-products are important to improve the performance of biofuel systems, either in terms of energy efficiency or GHG intensity;
- land use change scenarios exert a strong influence in the GHG intensity of biofuel systems;
- parameter uncertainty ranges for GHG intensities are higher than for ERenEf results;
- main sources of GHG uncertainty are soil emissions (carbon and nitrous oxide);
- under more conservative LUC scenarios (i.e. with lower carbon emissions from LUC), parameter uncertainty in GHG emissions overwhelms differences between co-product treatment approaches.

Concerning energy renewability efficiency results, it has been shown that:

- rapeseed oil has the highest energy renewability efficiency: depending on the co-product treatment approach, median RO ERenEf values range from 76% (substitution method) to 86% (mass allocation);
- RME and wheat-based bioethanol are intermediate in terms of non-renewable primary energy consumption throughout the life-cycle: median ERenEf values range from 55% (substitution) to 69% (mass) for RME, and between 34% (substitution) and 65% (mass) for bioethanol from wheat;

- sugar beet-based bioethanol has the lowest life-cycle energy efficiency: median ERenEf values range from 35% (substitution) to 47% (mass allocation) in the bioethanol pathway, and from -59% (substitution) to 48% (mass allocation) in the sugar pathway;
- conversion of bioethanol into its derivative bioETBE lowers the energy renewability of the life-cycle due to the use of (fossil) isobutylene feedstock;
- mass allocation generates the highest ERenEf results, which can be explained by the lower energy (LHV) and economic (market prices) parameters associated with biofuel co-products;
- the substitution method gives the lowest ERenEf results, due to low energy credits of co-product alternatives;
- parameter uncertainty ranges in ERenEf results are small.

An exception to the general remarks made above is the use of co-product glycerin as a replacement for synthetically-produced glycerin in the RME chain, which generates high energy credits and thus the highest ERenEf result. This option, however, represents a restricted market at the world and European level. Therefore, substitution of synthetic glycerin is more interesting as an academic scenario, showing the significant effect that the choice of a replaced product may have in the results, rather than representing an important and actual market scenario.

Concerning life-cycle GHG intensity, results show that sugar beet-based bioethanol has the lowest values when the GiA LUC scenario is considered, and the highest values with the CfA scenario. This is explained by the higher productivity (energy output per hectare, cf. section 6.4) of sugar beet compared to the other crops, which reduces the significant impact of soil carbon emissions due to LUC (both in the positive and negative directions, resp. GiA and CfA LUC scenarios). Finally, the other biofuel systems under analysis (RO, RME, and wheat-based bioethanol) show similar GHG intensities, with a slight advantage for the former.

6.4. EFFICIENCY IN LAND USE

Competition due to limitation of available land for growing energy crops justifies a comparison between biofuel systems in terms of land use efficiency. The energy output that can be achieved per hectare of cultivated land for each biofuel system addressed in this dissertation is shown in Fig. 6.28. The best use for arable land in terms of energy production is achieved with sugar beet if the entire yield is dedicated to bioethanol production. Energy outputs in this case more than triple the outputs of the remaining options[3]. Bioethanol production from sugar beet is therefore the best alternative if the decisive factor is final energy per surface area. Significant differences between alternative pathways in the sugar beet chain – bioethanol and sugar pathways – make sugar beet also the option with the lowest energy output when the sugar pathway is followed. Rapeseed oil, RME and wheat-based bioethanol are very similar in terms of energy output, with a small advantage for the latter.

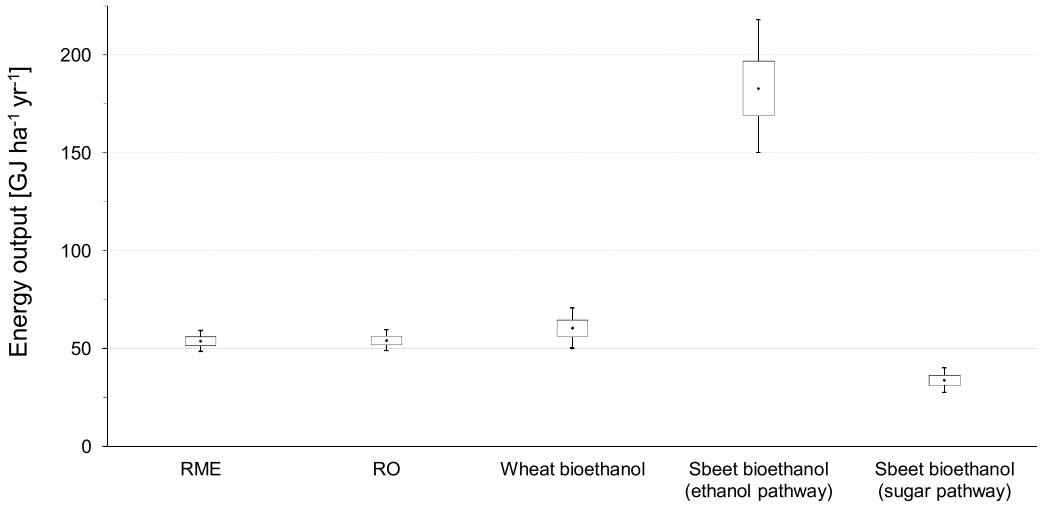


Fig. 6.28. Biofuel energy output per cultivated hectare dedicated to energy crops.

[3] Average biofuel yields per hectare of arable land: 1.5 t of RO and RME; 2.2 t of bioethanol (wheat); 6.8 t of bioethanol (sugar beet, bioethanol pathway); and 1.3 t of bioethanol (sugar beet, sugar pathway).

Figs. 6.29 to 6.31 show biofuel GHG emission savings over fossil fuels on a per hectare basis. Savings have been calculated excluding carbon emissions due to LUC. Because life-cycle GHG emission ranges per MJ are not very different for the analyzed biofuel systems (cf. Figs. 6.12, 6.15, 6.19, 6.24a, and 6.25a, for the CA LUC scenario), the GHG savings that can be realized per hectare follow the same trend of the energy outputs of Fig. 6.28:

- highest GHG savings per hectare are achieved with sugar beet cultivation and sugar beet processing into bioethanol;
- lowest GHG savings per hectare are associated with sugar beet dedicated to sugar production, with bioethanol as a co-product;
- RME, RO and wheat-based bioethanol present similar GHG savings, slightly lower in the former due to slightly higher life-cycle GHG intensity;
- GHG savings are lower with the substitution method in the RME, RO and wheat-based bioethanol chains (the exception is glycerin displacing synthetic glycerin in the RME chain), due to low GHG credits associated with substitution.

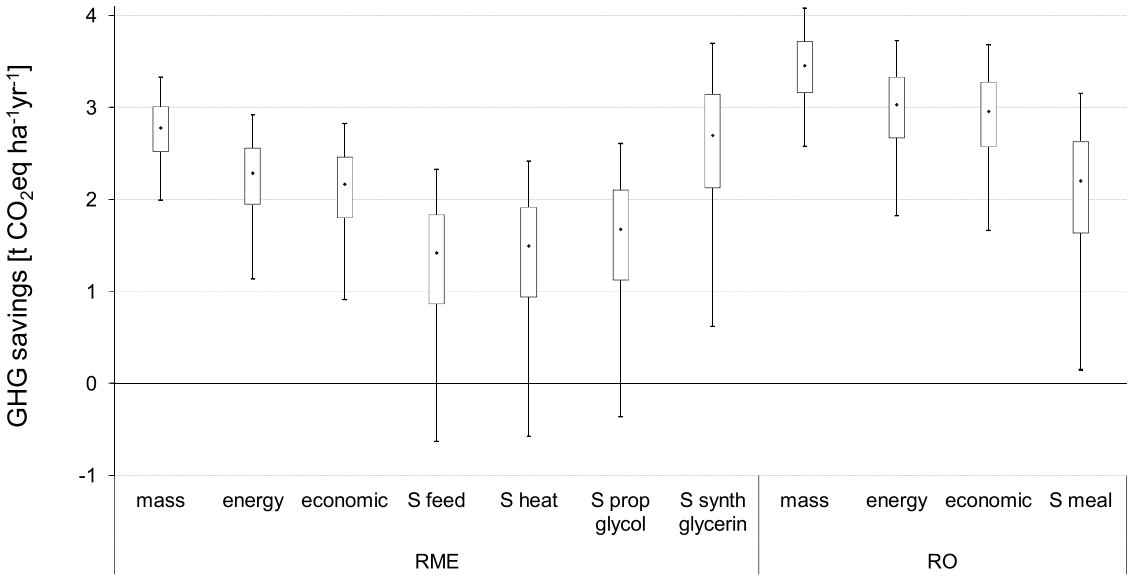


Fig. 6.29. GHG emission savings on a per hectare basis when RME and RO displace fossil diesel (S: substitution method).

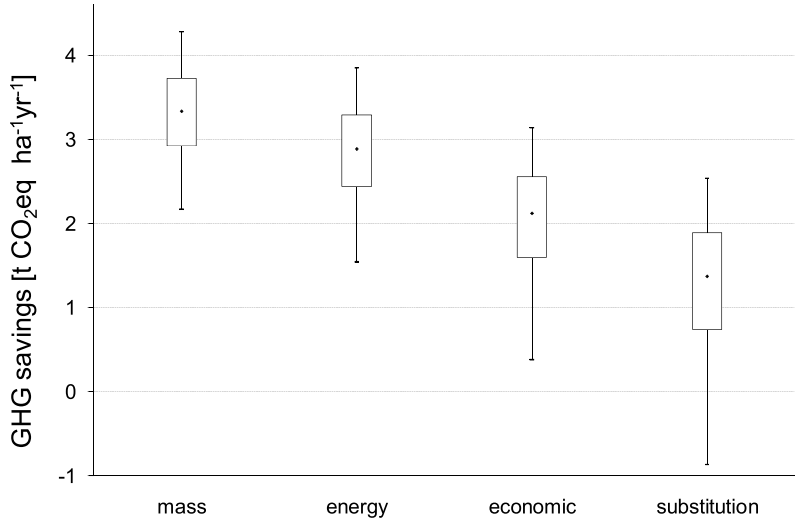


Fig. 6.30. GHG emission savings on a per hectare basis when wheat-based bioethanol displaces gasoline.

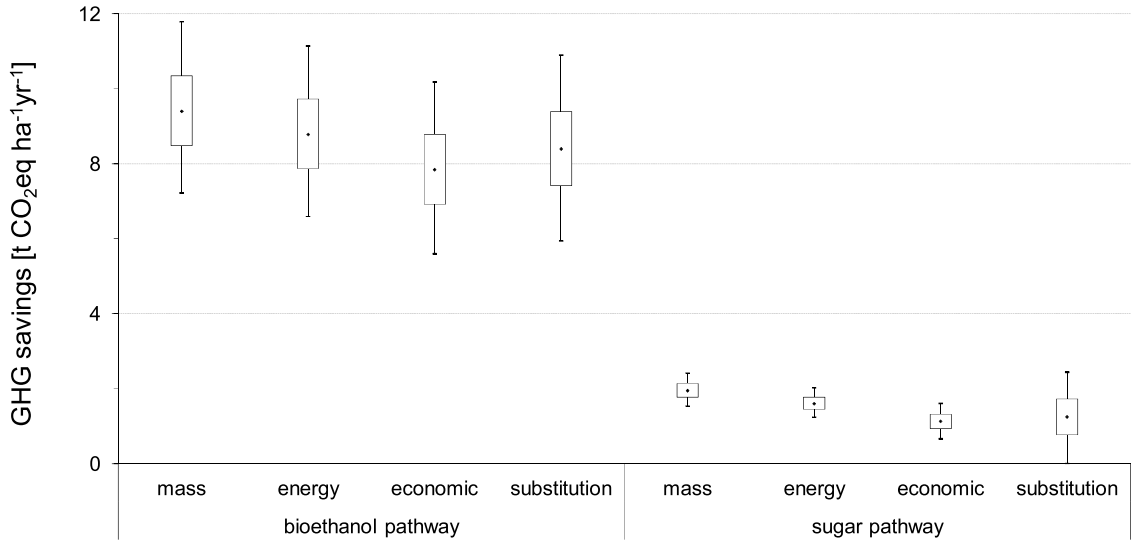


Fig. 6.31. GHG emission savings on a per hectare basis when sugar beet-based bioethanol displaces gasoline.

6.5. CARBON PAYBACK TIME

As demonstrated in the previous section, biofuel production systems may have considerably higher GHG emissions than equivalent fossil fuels (fossil diesel and gasoline) if the most severe land use change scenario (improved grassland converted to cropland, GiA) is considered. On the other hand, biofuel life-cycle GHG emissions not accounting for LUC issues are on average lower than fossil fuel emissions, indicating that biofuel production can gradually compensate the LUC emissions that mostly occur in the first years after land use conversion.

If the aim is to achieve GHG savings by replacing fossil fuels with biofuels, it is therefore important that the period of time a specific biofuel system takes to compensate potential LUC emissions be estimated ("carbon payback time", CPT). In the following paragraphs, calculated CPT values for the biofuel chains addressed in this dissertation are presented. The most severe LUC scenario (GiA) has been selected for the calculation. The other LUC scenarios (CA and CfA) have lower soil carbon emissions (or even carbon sequestration) and contribute mostly to biofuel GHG savings over fossil fuels from the 1st year after land use conversion. These scenarios are thus not relevant for the analysis. Table 6.1 details an example of CPT calculation for rapeseed oil displacing fossil diesel.

Table 6.1. Carbon payback times (CPT) for rapeseed oil displacing fossil diesel (mass allocation; LUC scenario: GiA).

Percentile	**5th**	**25th**	**50th**	**75th**	**95th**
GHG emissions without LUC [a] [g CO_2eq MJ^{-1}]	12.3	14.8	17.3	21.2	32.2
Fossil Diesel emissions [g CO_2eq MJ^{-1}]	Normal distribution			(μ = 82; σ = 3)	
GHG emission savings relative to FD [t CO_2eq ha^{-1} yr^{-1}]	2.58	3.16	3.45	3.71	4.08
ΔC_{LUC} [b] [t CO_2eq ha^{-1}]	152.7	184.1	206.4	229.0	262.0
CPT [yr]	42.3	52.3	60.2	69.2	86.9

[a] Greenhouse gas emissions of RO without including changes in soil carbon stocks due to LUC.

[b] ΔC_{LUC}: soil carbon stock change due to LUC.

Fig. 6.32 shows carbon payback times calculated for rapeseed oil (RO) and rapeseed methyl ester (RME) displacing fossil diesel. Regardless of the co-product treatment approach, it takes several decades to pay off in GHG terms: calculated carbon payback times range from 52 to 120 years (25th and 75th percentiles) for RO and from 65 to 190 years for RME. In particular, combinations of input values that result in higher GHG emissions (and thus lower GHG savings over petroleum diesel) may lead to very high CPT values, beyond 250 (RO) and 500 (RME) years. Main conclusion from Fig. 6.32 is that promotion of RO or RME associated with conversion of improved grasslands is always a non-GHG saving measure over petroleum diesel in the short- to mid-term. With the exception of specific substitution options for glycerin under most favorable conditions of parameter uncertainty, promotion of RO or RME as a replacement for petroleum diesel using improved grasslands is always counter-productive to avoid climate change consequences in the near-term.

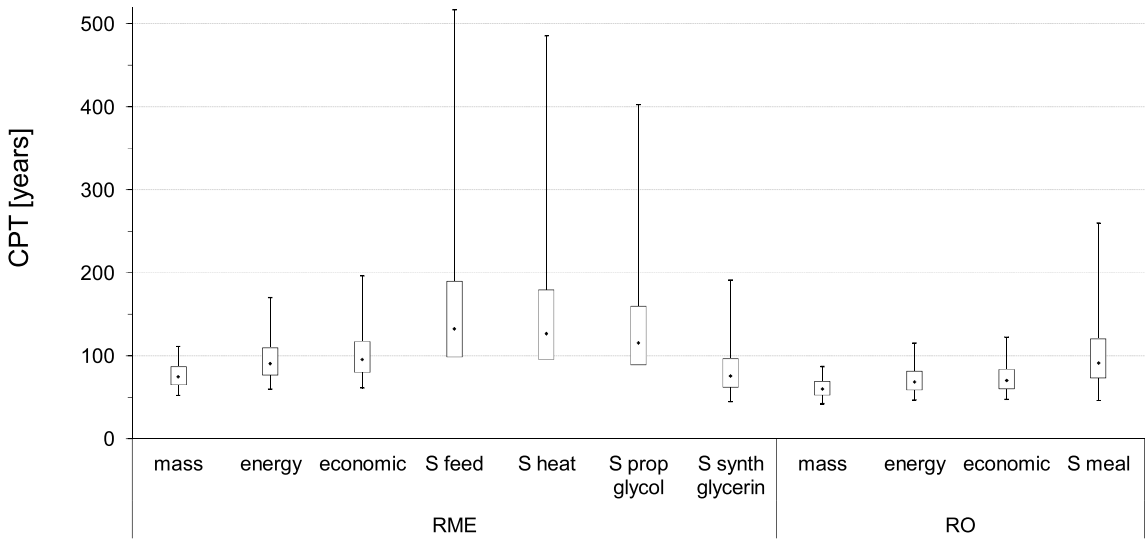


Fig. 6.32. Calculated carbon payback times (CPT) when RME and RO displace fossil diesel. LUC scenario: GiA. Substitution method (S): four different applications for glycerin (feed; heat; propylene glycol; synthetic glycerin); rape meal displacing soy meal. Negative 5th percentile points of the substitution method have been removed due to physical inconsistency.

Calculated carbon payback times when wheat-based bioethanol displaces gasoline (Fig. 6.33) show that conversion of improved grassland to wheat cultivation is not a GHG saving measure in the short- to mid-term: Fifty years after land use conversion, GHG savings from biofuel production still do not compensate carbon emissions from land use change. As regards allocation, it takes on average 62 to 95 years for bioethanol GHG savings to equalize and thus pay back LUC emissions. Concerning the substitution method, the higher uncertainty ranges of GHG emissions result in a wide range of calculated CPT values. Comparing Figs. 6.32 and 6.33 shows that RME and wheat-based bioethanol have similar carbon payback times.

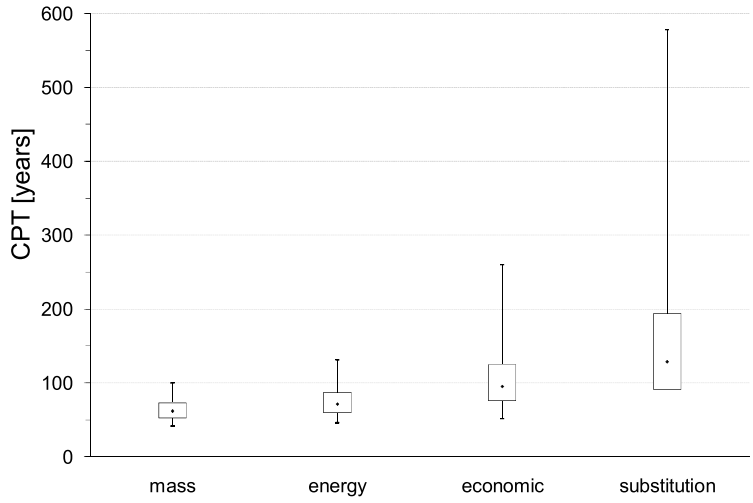
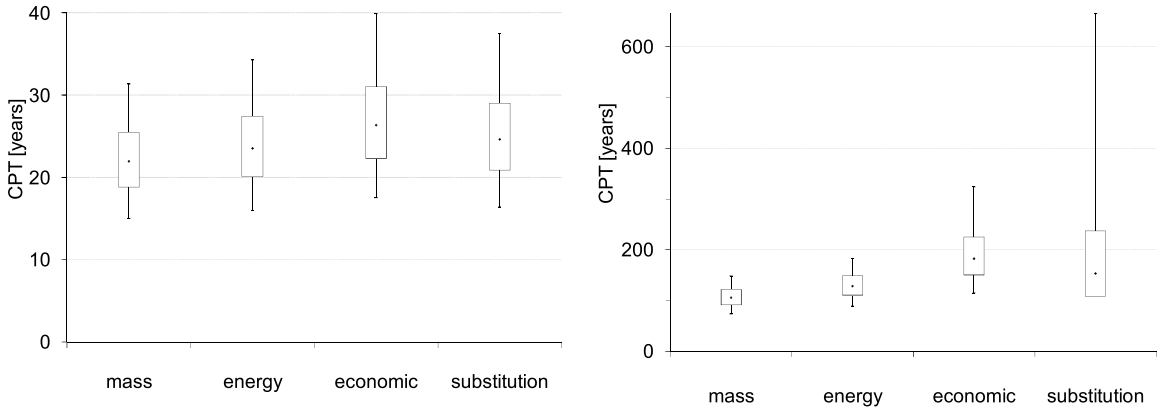


Fig. 6.33. Calculated carbon payback times (CPT) when wheat-based bioethanol displaces gasoline. LUC scenario: GiA. The negative 5th percentile point of the substitution method has been removed due to physical inconsistency.

Concerning sugar-beet based bioethanol, Fig. 6.34 shows the disparity between the bioethanol and sugar pathways. The former shows relatively low carbon payback periods (between 20 and 30 years, on average), whereas for the latter it takes more than 100 years of bioethanol displacing gasoline to compensate for the increased emissions associated with LUC. This can be explained by the higher level of life-cycle GHG emissions allocated to bioethanol in the sugar pathway. It can be concluded that

bioethanol from sugar beet is a good option as a GHG saving measure over fossil gasoline, even in the most severe scenario of land use change. It is also the best option amid the alternatives analyzed in this dissertation.



(a) (b)

Fig. 6.34. Calculated carbon payback times (CPT) when sugar beet-based bioethanol displaces gasoline: (a) bioethanol pathway; (b) sugar pathway. LUC scenario: GiA. The negative 5th percentile point of the substitution method has been removed due to physical inconsistency.

6.6. SENSITIVITY ANALYSIS TO IMPORTANT PARAMETERS

6.6.1. Introduction

In this section, a sensitivity analysis to important aspects influencing the life-cycle results of biofuel systems is presented. Assessed issues include type of soil and climate region, agricultural productivity levels, and application of animal manure instead of mineral fertilizer. Moreover, model outcomes are compared using different time frames for annualization of LUC emissions and calculation of global warming potentials (uncertainty due to choices). Wheat-based bioethanol is chosen to illustrate the implications in each case, but most of the analysis also applies to other biofuel systems. In particular, when GWPs are estimated for different time horizons, GHG intensity results may vary depending on the share of CO_2, CH_4 and N_2O in the life-cycle. Nonetheless, the biofuel systems under analysis show similar shares.

6.6.2. Type of soil and climate region

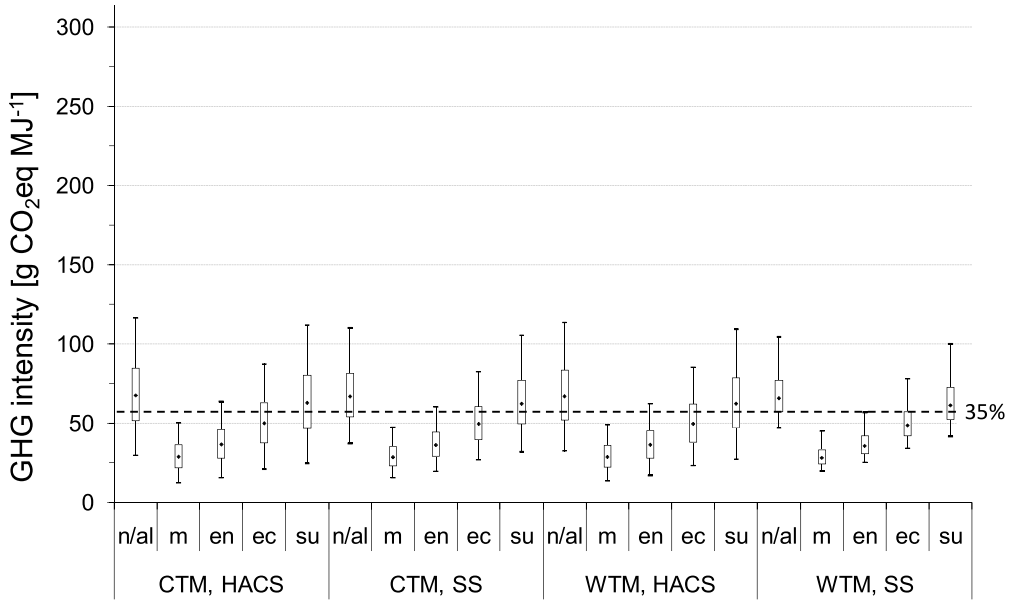
As already discussed in chapter 5, the type of soil and climate region selected for cultivation of energy crops are aspects that affect the GHG balance of biofuel systems and thus their performance over fossil fuels. An illustrative example for wheat-based bioethanol including two different LUC scenarios – GiA (improved grassland to cropland) and CA (cropland to cropland) – is used to show the implications of soil type and climate region in the GHG intensity of biofuel systems (Fig. 6.35).

Comparing Figs. 6.35a (CA LUC scenario) and 6.35b (GiA LUC scenario), it can be seen that:

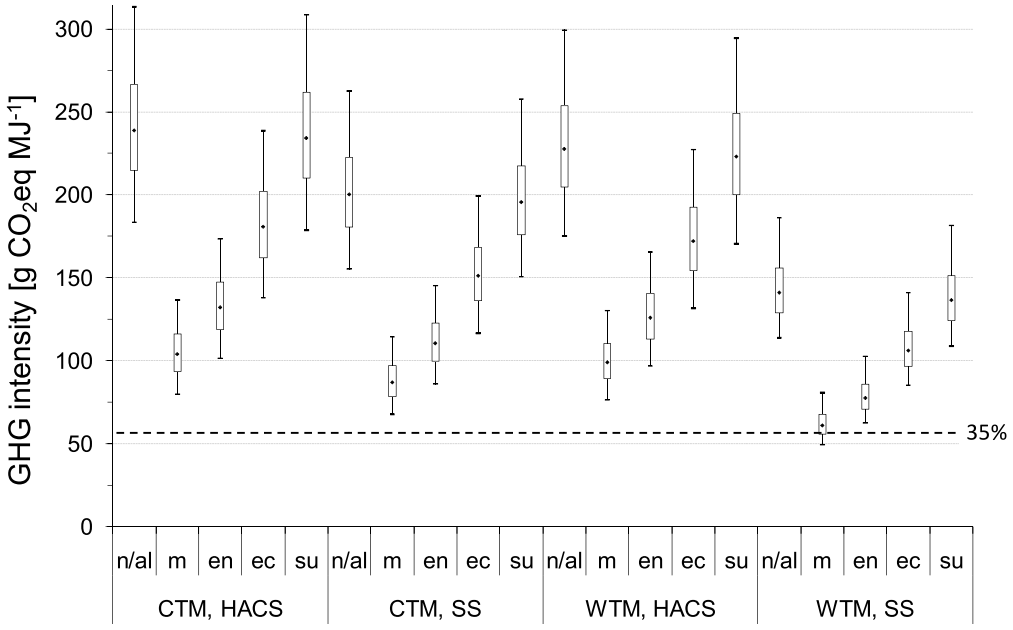
- there are no effective differences between soil types or climate regions in the most conservative LUC scenario (Fig. 6.35a) and GHG intensities mainly contribute to GHG savings over fossil fuels;
- in the GiA scenario, differences are shown among soil types and climates. The combination of warm temperate moist climates with sandy soils (WTM, SS), in particular, has low GHG emissions, even lower than fossil fuel emissions when mass or economic allocation approaches are used.

Moreover, Figs. 6.35a and 6.35b show that:

- in the most conservative LUC scenario (CA, Fig. 6.35a), the median value for ΔC_{LUC} is zero. Thus, variation in GHG results for different soil types and climate regions is mainly seen through different uncertainty ranges;
- in the GiA scenario, net carbon emissions associated to LUC are amplified in soils/climates for which soil carbon content is high (cf. Table 5.1), namely HACS soils and both types of climates;
- although there are differences between co-product treatment approaches in the CA LUC scenario, parameter uncertainty ranges partly compensate for these differences;
- conversely, for higher LUC emissions (GiA LUC scenario), differences between co-product treatment approaches are very clear, even when parameter uncertainty is taken into account.



(a)



(b)

Fig. 6.35. Life-cycle GHG intensity of wheat-based bioethanol: (a) CA LUC scenario; (b) GiA LUC scenario. (CTM: cold temperate moist climate; WTM: warm temperate moist climate; HACS: high activity clay soil; SS: sandy soil).

6.6.3. Agricultural yield

This section shows the implications of different agricultural (wheat) yields in the GHG intensity of wheat-based bioethanol. Two distinct ranges have been considered: (i) a range for the top-3 European wheat producers; and (ii) a range for the 4th and 5th (and other low-productivity) European producers. Table 6.2 gathers data concerning major European wheat producers in recent years and the probability distributions used in this assessment.

Table 6.2. Agricultural productivities of major European wheat producers (based on data from FAOSTAT 2010).

EU Ranking (production)	Country	Wheat Productivity 2005-2009 [t ha^{-1}]			Data used [t ha^{-1}]
		Average	Min	Max	
1st	France	6.91	6.25	7.45	Normal distribution (μ = 7.40; σ = 0.74)
2nd	Germany	7.51	6.96	8.09	
3rd	United Kingdom	7.89	7.22	8.28	
4th	Poland	3.87	3.24	4.17	Normal distribution (μ = 3.75; σ = 0.25)
5th	Italy	3.64	3.41	3.87	

Fig. 6.36 shows the variation in GHG intensity of wheat-based bioethanol for the CA LUC scenario. In addition to a better efficiency in the use of land, higher agricultural yields have a significant effect in decreasing the GHG intensity of the life-cycle. Main conclusion is that agricultural yield is a key parameter in the life-cycle GHG results of biofuel systems.

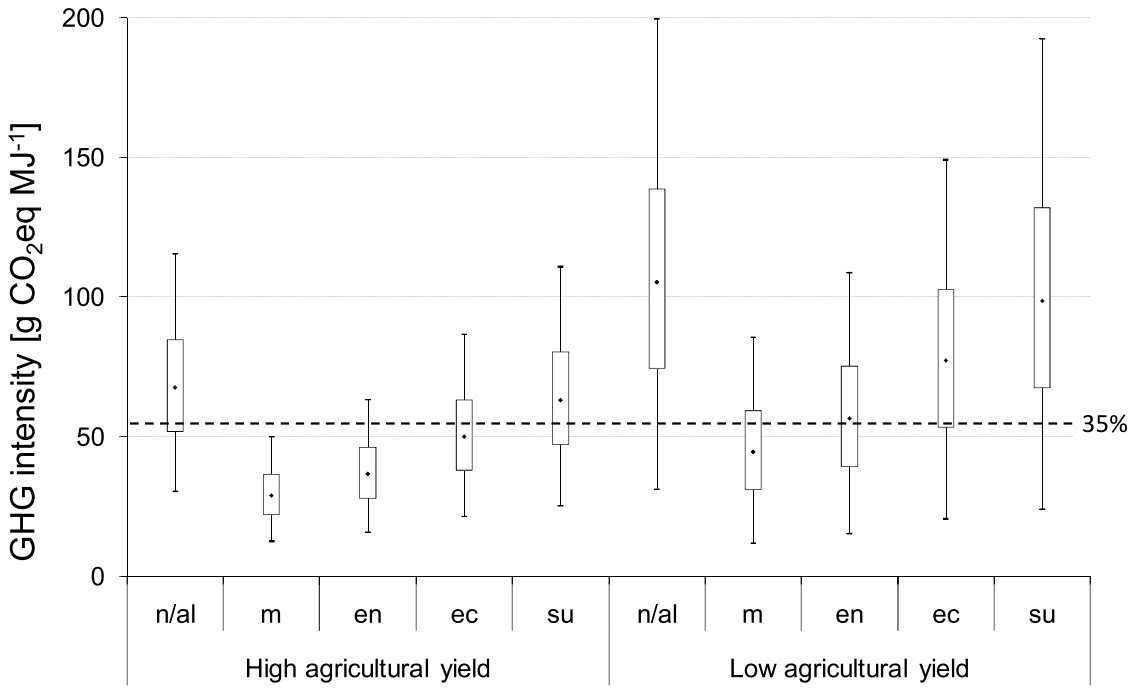


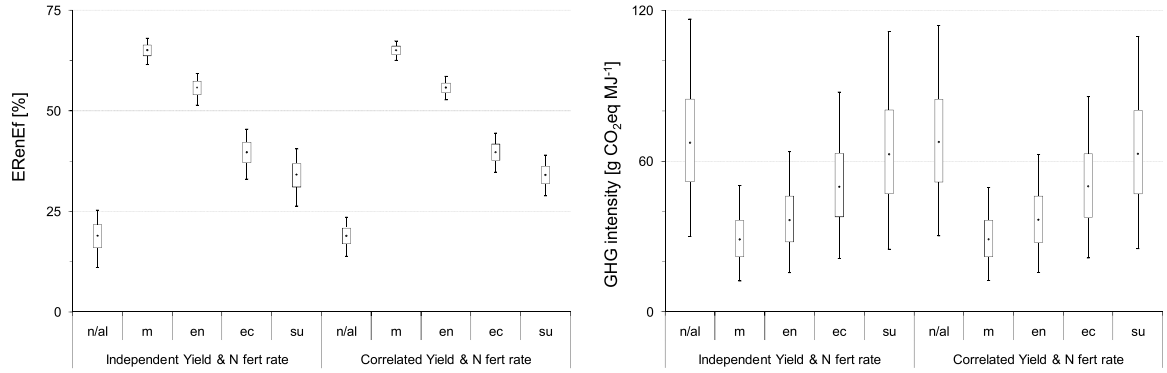
Fig. 6.36. Life-cycle GHG intensity of wheat-based bioethanol for high- and low-agricultural-yield producers.

6.6.4. Correlation between fertilizer application rate and agricultural yield

This section shows the implications of including a correlation between fertilizer application rates and wheat productivity in the life-cycle model of wheat bioethanol. Kuesters and Lammel (1999) present a relationship between wheat grain yield [t ha^{-1}] and the application rate of nitrogen fertilizer [kg N ha^{-1}] for different wheat growing conditions, namely good, medium and poor conditions in terms of soil type and weather. Based on data from Kuesters and Lammel (1999), a correlation coefficient has been calculated between wheat yield and N fertilizer application rate (r=0.95). This correlation has been included in the Monte Carlo simulation to restrain the choice of values for the correlated parameters.

The implications of using a correlation coefficient instead of maintaining independence between wheat yield and N fertilizer application rate in the life-cycle model are shown in Fig. 6.37. It can be observed that the variation in results between independent and correlated parameters is only small (ERenEf results) or even insignificant (GHG intensity).

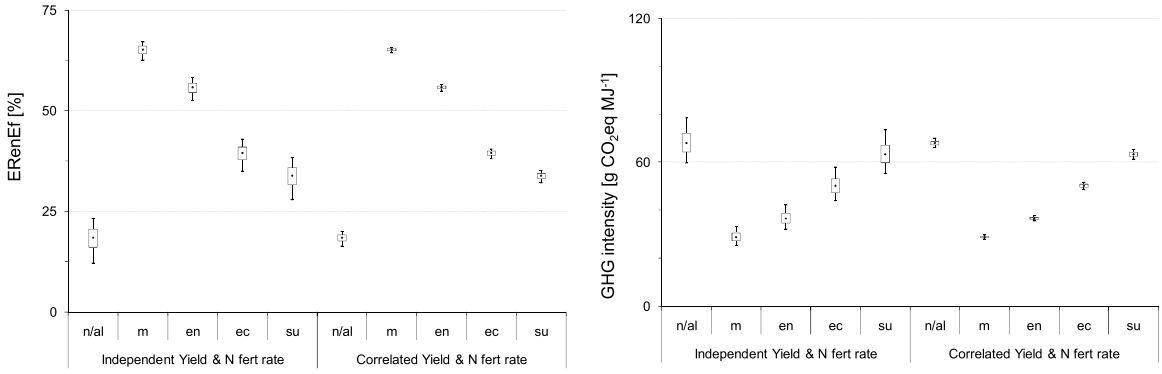
This can be explained by the fact that Monte Carlo simulation takes into account the probability distributions of all the parameters at once and thus the contribution of each parameter may only provide a small variation in the range of results. The difference between ERenEf and GHG intensity results is explained by the strong relationship between the amount of N fertilizer used and the energy consumption in the life-cycle.



(a) (b)

Fig. 6.37. Implications of correlation between N fertilizer application rate and agricultural yield in the (a) energy renewability efficiency and (b) GHG intensity of wheat-based bioethanol (LUC scenario: CA).

In order to understand the effect that other uncertain parameters have in the life-cycle modeling of wheat-based bioethanol, a simulation has been conducted in which all the parameters have been frozen in their average values, except N fertilizer application rate and wheat yield. Results are shown in Fig. 6.38. The inclusion of a correlation coefficient reduces the uncertainty ranges for both ERenEf and GHG intensity results, irrespective of the co-product treatment approach considered. This result shows the importance of taking into account correlation between parameters in the life-cycle model, even though in this example the inclusion of additional uncertain parameters masks the effect of correlation, as shown in Fig. 6.37. An important conclusion is: as further investigation provides more accurate data, the overall uncertainty ranges may be reduced and correlation issues may become increasingly important in the life-cycle model.



(a) (b)

Fig. 6.38. Implications of correlation between N fertilizer application rate and agricultural yield in the (a) energy renewability efficiency and (b) GHG intensity of wheat-based bioethanol (LUC scenario: CA). All parameters frozen at their average values, except N fertilizer application rate and wheat yield.

Although mathematical correlation between agricultural yield and N fertilizer application rate may be an important issue, this correlation has not been included in the biofuel life-cycle models addressed in this dissertation. The reason is twofold:

- the correlation exists for given conditions when growing energy crops. For varying and uncontrollable conditions (e.g. weather), it is possible that similar grain yields are obtained for very different fertilizer application rates, as shown by Kuesters and Lammel (1999);
- under large uncertainty ranges, the effect of correlation is not important, as shown in Fig. 6.37.

6.6.5. Application of animal manure

The baseline scenario in this dissertation assumes the application of mineral fertilizer in order to meet the crop requirements of major nutrients, although it is also possible to use organic fertilizer. This section investigates the implications on the life-cycle GHG intensity of wheat-based bioethanol of substituting part of the mineral fertilizer with animal manure.

Elsgaard (2010) evaluated the GHG emissions of wheat and rapeseed cultivation in Denmark. One of the scenarios assumed that crops were partially amended using animal slurry with an N fertilizer efficiency of 75%. Chirinda et al. (2010) also compared conventional (mineral fertilizer) and organic (pig slurry) fertilization systems in wheat cultivation in Danish soils, but with lower application rates (in terms of N fertilizer equivalents) in the latter. In this dissertation, the N fertilizer application rate of the baseline scenario is half substituted with animal manure in the organic fertilizer scenario, according to Table 6.3. Concerning field emissions, Chirinda et al. (2010) found no significant differences in N_2O emissions between conventional and organic cropping systems. It is therefore assumed that the use of manure results in field emissions equal to those of mineral fertilizer application.

Table 6.3. Scenarios concerning N fertilization in wheat cultivation.

Scenario	Fertilization	Distribution	Data used [kg N ha^{-1}]
Baseline	Conventional (mineral fertilizer)	Normal	(μ = 160; σ = 20)
Organic	50% mineral fertilizer +	Normal	(μ = 80; σ = 14.1)
	50% animal manure	Normal	(μ = 80; σ = 14.1)

Finally, because the storage of animal slurry is associated with CH_4 and N_2O emissions, a sensitivity analysis is performed to assess the contribution of management and storage of slurry in the life-cycle GHG intensity of wheat-based bioethanol. Emissions of this step are estimated based on Elsgaard (2010).

Fig. 6.39 shows that the use of animal manure significantly contributes to decrease the GHG intensity of bioethanol from wheat. This reduction can be explained by the contribution of manure to raise the carbon content of soils, in accordance with the guidelines of EC (2010). The carbon exchange due to land use change in this case is actually a carbon sequestration process. Handling and storage of animal manure represents on average 10g CO_2eq MJ^{-1}, which is not relevant when compared to the benefits of manure application.

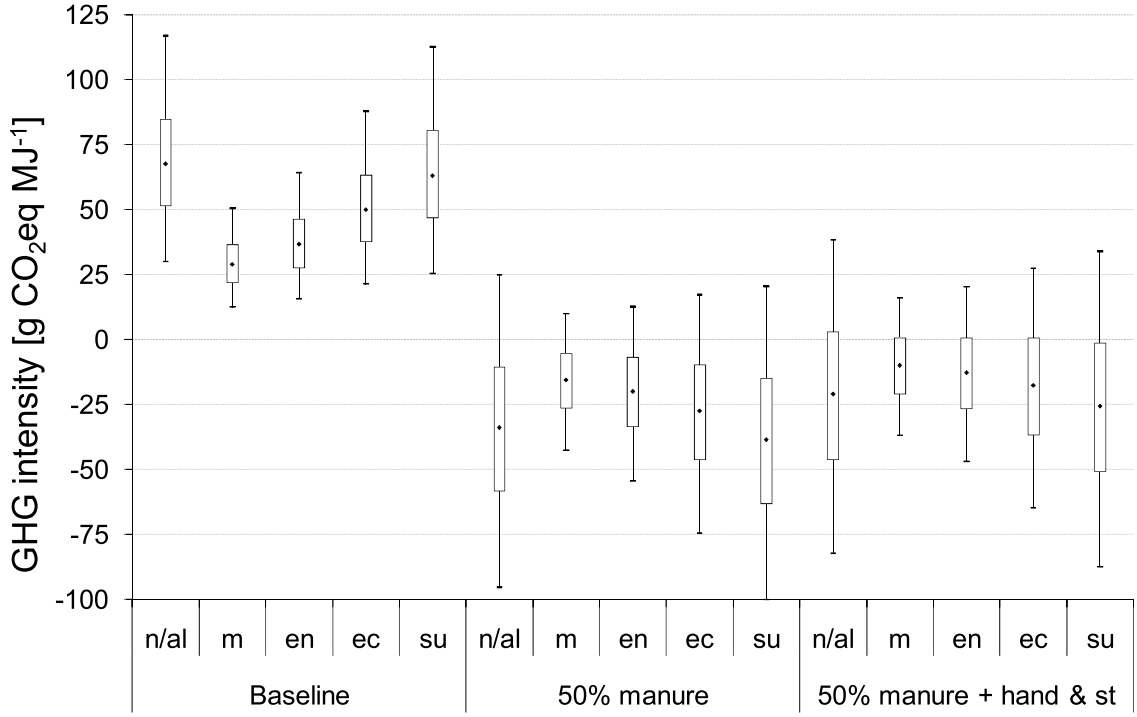


Fig. 6.39. Implications of using animal manure in the life-cycle GHG intensity of wheat-based bioethanol (baseline LUC scenario: CA; hand & st: handling and storage of animal manure).

6.6.6. Time frame for annualization of LUC emissions

Implications in the GHG intensity of biofuels of the time frame chosen for annualization of soil emissions from LUC are shown in Fig. 6.40. Three common periods have been considered, as discussed in chapter 2: twenty; ten; and thirty years. The most severe scenario in terms of carbon emissions from land use change (improved grassland to cropland, GiA) is selected because it maximizes variation between different time frames. Fig. 6.40 shows the significant difference in life-cycle GHG emissions of opting for alternative time frames. For example, choosing a 30-year period for amortization of LUC emissions is sufficient to reduce the life-cycle GHG intensity of wheat-based bioethanol to gasoline levels (mass allocation), even acknowledging that this is the most severe LUC scenario. Conversely, for the same co-product treatment approach but a 10-year annualization period, the GHG intensity of wheat-based bioethanol more than doubles gasoline levels, on average. Finally, differences between time frames are smaller when allocation approaches are used, as well as for less severe LUC scenarios.

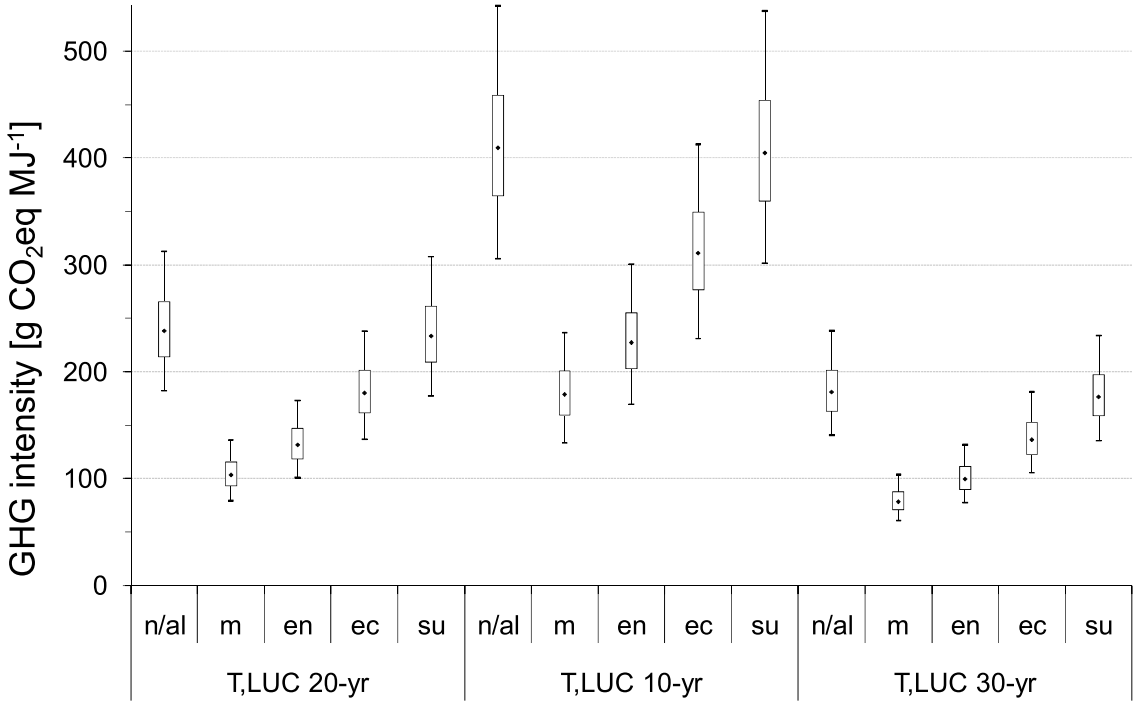


Fig. 6.40. Life-cycle GHG intensity of wheat-based bioethanol for different annualization periods of LUC emissions (LUC scenario: GiA).

6.6.7. Time horizon for GHG global warming potentials

The influence of characterization factors in Life Cycle Impact Assessment can be considered by running different options as scenarios (ISO 14044:2006). Thus, in addition to a baseline scenario in which the global warming potentials of GHGs are computed for a time horizon of 100 years, the GHG intensity of wheat-based bioethanol has been calculated using GWPs for other time horizons, namely 20 and 500-years (Fig. 6.41). Probability distributions have been considered for the global warming potentials of nitrous oxide and methane (Table 6.4). Carbon dioxide, being the reference gas in terms of global warming, has a unitary GWP for all time horizons.

Results show that 500-yr GHG emissions are lower than 100-yr GHG emissions due to a significantly lower GWP of nitrous oxide (average values of 153 and 298 kg CO_2eq, for 500- and 100-yr respectively). Moreover, uncertainty ranges for a 500-yr timeframe are slightly narrower than corresponding 100-yr values, because of the lower uncertainty in the estimation of GWP_{N2O}. On the other hand, calculated GHG intensity 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 wheat-based bioethanol, the implications of GWP_{CH4} variation between different time horizons are not significant. When the 35% threshold of the RED directive (EPC 2009) for GHG emission savings over fossil fuels is considered, Fig. 6.41 (next page) shows that there is more than 50% probability that the target is not achieved when the substitution method is used (GWP 100-yr), whereas with the other method indicated in the directive (energy allocation) GHG savings are almost certainly above the 35% target.

Table 6.4. Probability distributions for the global warming potentials (GWP) of methane and nitrous oxide (calculated on the basis of IPCC 2007).

PARAMETER	distribution	μ	σ
GWP-CH_4 [g CO_2eq]			
100-yr	Normal	25	5.32
20-yr	Normal	72	15.3
500-yr	Normal	7.6	1.6
GWP-N_2O [g CO_2eq]			
100-yr	Normal	298	63.4
20-yr	Normal	289	61.5
500-yr	Normal	153	32.5

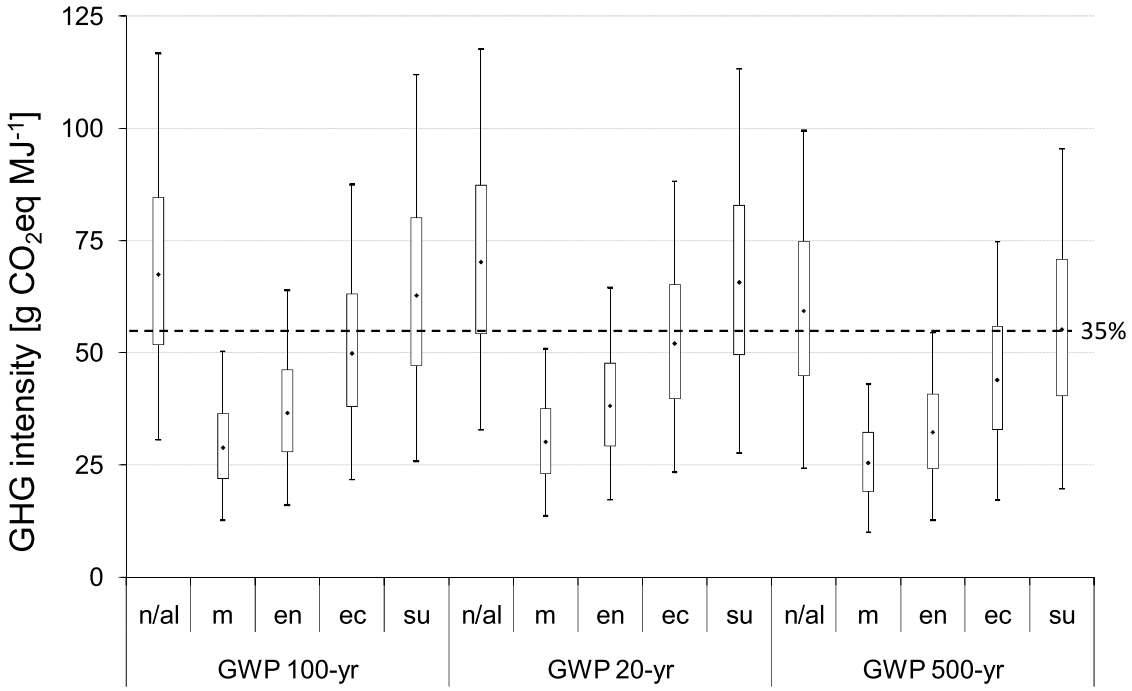


Fig. 6.41. Life-cycle GHG intensity of wheat-based bioethanol under different time horizons for GWPs (baseline LUC scenario: CA).

7. Concluding Remarks

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7. CONCLUDING REMARKS

7.1. SUMMARY OF FINDINGS

Renewable energy sources, including biofuels, are expected to play an increasingly important role in coming years, as we search for ways to improve security of energy supply and reduce global warming (EPC 2009; EPA 2010). Nevertheless, significant disagreement and controversies exist regarding the actual benefits of biofuels displacing fossil fuels. Several studies addressing energy and environmental issues in the life-cycle of biofuels show varying and sometimes contradictory outcomes, even for the same biofuel and pathway, which emphasizes the need to identify and improve the knowledge of the main drivers for the differences between studies (and also within specific studies). Are they due to different methodological procedures (or modeling choices), data or production conditions?

Even though a few life-cycle assessment studies take into account uncertainty and variability issues, they usually consider the treatment of uncertainty as an appendix. Furthermore, several review studies show that important aspects for the GHG balance of biofuels have not been taken into account, even in recent biofuel life-cycle studies.

Acknowledging the sustainability concerns on biofuels at the international agenda, robust life-cycle modeling approaches incorporating uncertainty are essential to improve the transparency and reliability of life-cycle studies and better support decisions on whether or not to support specific biofuel pathways. Against this background, the following research questions have been formulated (cf. chapter 1):

I. What drives the differences and sometimes contradictory conclusions between life-cycle studies, even for the same biofuel pathway?
II. How to develop life-cycle models for biofuel systems incorporating uncertainty?
III. How uncertain are the energy and GHG emission results from European biofuel (biodiesel and bioethanol) life-cycle studies?
IV. Given the uncertainty ranges, is it possible to ensure that biofuels are really delivering energy and GHG savings over displaced petroleum fuels? And to what extent?
V. What direction should research take to improve the robustness of biofuel life-cycle studies?

This dissertation addressed these questions as follows. Question I is thoroughly discussed in chapter 3 and question II is analyzed in chapter 4. Chapters 3, 5, and 6 give answers to question III. Question IV is responded in chapters 3 and 6. Finally, question V is addressed in section 7.2 of this chapter. The main findings of the chapters are summarized in the following paragraphs.

Firstly, an introduction to the theoretical grounds of uncertainty analysis in the life-cycle assessment of biofuels is presented in chapter 2, which is a background chapter for the remainder of the dissertation. Secondly, a comprehensive review of life-cycle studies published in recent years for biodiesel (from rapeseed) and bioethanol (from wheat and sugar beet) in Europe is presented in chapter 3. This review also provides an understanding of how methodological and data limitations of studies can be assessed and overcome. Studies have been compared in terms of non-renewable primary energy requirement and GHG intensity. A detailed description of relevant aspects, including modeling choices, has been included to identify the main causes for the high variability

of results. It has been demonstrated that there is a correlation between key modeling issues addressed by surveyed life-cycle models, namely soil emissions, and the GHG intensity of biofuels. Moreover, it has been demonstrated that taking into account soil emissions in biofuel life-cycle assessments negates the correlation between non-renewable energy inputs and GHG emissions presented by most former studies. Important recommendations for life-cycle studies are drawn from this survey, namely the need to conduct uncertainty importance analysis to highlight areas in which an improved understanding is needed, and the call for a better understanding of the importance of different types of uncertainty.

Thirdly, this dissertation describes first-generation biofuel systems in Europe, with special emphasis on modeling issues, namely the treatment of multifunctionality and data uncertainty (chapter 4). Five biofuel chains have been addressed: vegetable oil and biodiesel from rapeseed; bioethanol from wheat and sugar beet, and its derivative bioETBE. Extensive data collection has been conducted to build life-cycle inventory tables and assist in the selection of probability density functions capturing parameter uncertainty.

One of the most important issues affecting the GHG balance of biofuel systems, as demonstrated in the systematic review of chapter 3, is the inclusion of soil carbon emissions from direct land use change. A thorough discussion on this aspect has been conducted (chapter 5). In particular, different approaches for estimation of soil carbon fluxes have been addressed and sensitivity analyses on several important variables have been performed, namely concerning different agricultural practices (soil management and carbon inputs to soil), climate regions, and types of soil. Large differences between scenarios in LUC modeling have been shown. In particular, conversion of grasslands to energy crops is the scenario with higher GHG emissions from dLUC, with the exception of grasslands in severely degraded lands, which are particularly poor in terms of soil carbon stock, and thus show benefits when converted to energy crops. Moreover, calculated values show that the uncertainty of soil carbon fluxes when the error ranges of IPCC (2006) guidelines are taken into account is very high, which calls for further research to gradually narrow this source of uncertainty. High activity clay soils in cool or warm temperate moist climates and sandy soils in cool temperate moist climates have

the highest standard soil carbon stocks and thus are linked to higher carbon fluxes, either as net emissions or sequestration.

Fourthly, a framework to incorporate uncertainty in the life-cycle GHG emission and renewability assessment of biofuels has been implemented to representative first-generation biofuel systems brought from the European context, which enables comparison between biofuels and also against displaced petroleum fuels. Results are discussed in chapter 6. Main conclusions can be summarized as follows:

- land use change dominates the GHG intensity of biofuels, but there is a high level of uncertainty;
- GHG emissions show higher uncertainty ranges than energy requirement for all biofuel systems, which is mainly due to carbon emissions from LUC and soil N_2O emissions;
- for LUC scenarios with lower carbon emissions, parameter uncertainty in GHG emissions overwhelms differences between co-product treatment approaches;
- optimum use of co-products is required to improve the energy efficiency and GHG intensity of biofuels;
- in terms of energy renewability efficiency, RO has the highest results (ERenEf values from 58% to 88%), followed by RME and wheat-based bioethanol. Sugar beet-based bioethanol has the lowest life-cycle energy efficiency (from 24% to 49%);
- the energy-intensive process of bioETBE synthesis brings the overall energy renewability efficiency of bioETBE to negative values, very close to gasoline´s ERenEf;
- in terms of life-cycle GHG intensity, sugar beet-based bioethanol (bioethanol pathway) has the lowest GHG emissions when the GiA LUC scenario (most severe in terms of soil carbon emissions) is considered, and the highest emissions with the CfA LUC scenario (which is associated with soil carbon sequestration). This is explained by the higher productivity (energy output per hectare) of sugar beet compared to the other crops;

- the other biofuel systems under analysis (RO, RME, and wheat-based bioethanol) show similar GHG intensities, with a slight advantage to the former;
- moving from bioethanol to bioETBE production reduces the uncertainty range of GHG emissions, and may increase or decrease calculated median values depending on the magnitude of bioethanol life-cycle emissions;
- when sugar beet is processed for sugar production, large variations between co-product approaches are shown, which are explained by the high mass share of sugar in this pathway and the substitution credits considered for sugar;
- conversion of full- or low-tillage croplands to energy crops results in biofuel life-cycle GHG emissions lower than equivalent fossil fuel emissions; the exception is the use of the substitution method in the RME, wheat, and sugar beet (sugar pathway) chains, with emissions that may overcome fossil fuel emissions;
- concerning land use efficiency, bioethanol from sugar beet presents the highest performance, with an energy per hectare output that more than triples the outputs of the other biofuel chains;
- calculated carbon payback times (CPT) for the LUC scenario of (improved) grassland conversion to energy crops show that this option does not contribute to GHG savings over fossil fuels in the short- to mid-term. CPT values are always above 50 years. The exception is sugar beet-based bioethanol (bioethanol pathway) with CPT results in the range of 20 to 30 years.

Finally, a sensitivity analysis to important aspects influencing the life-cycle results of biofuel systems has been conducted (section 6.6), namely type of soil and climate region, agricultural productivity levels, application of animal manure instead of mineral fertilizer, and time frames for annualization of LUC emissions and calculation of global warming potentials. It has been concluded that

- LUC scenarios with lower soil carbon emissions do not show any significant difference between soil types and climate regions, as opposed to scenarios with higher soil carbon exchange;

- agricultural yield is an important parameter affecting the GHG balance of biofuels; more and better data to model the dependencies between agricultural yields and other variables is therefore desired;
- the use of animal manure instead of mineral fertilizer significantly reduces the GHG intensity of biofuels;
- opting for different time frames for annualization of LUC emissions has a significant effect in the life-cycle GHG emissions of biofuels, which more than double when a 10-year period is chosen instead of 30-years;
- there is not a significant difference in the GHG intensity of biofuels under different time horizons for GWP calculation. Results are slightly lower with a 500-year time frame, due to a significantly lower GWP of nitrous oxide, whereas for 20- and 100-years results are similar.

The relevance of addressing uncertainty issues in biofuel life-cycle studies instead of using average (deterministic) approaches has been demonstrated. Following the methodology described in this dissertation, both the overall uncertainty and the relative importance of different types of uncertainty can be assessed.

As a closing remark for this dissertation:

Robust life-cycle modeling approaches incorporating uncertainty are essential to improve the transparency and reliability of life-cycle studies and better support decisions on whether or not to support specific biofuel pathways.

7.2. RECOMMENDATIONS FOR FUTURE RESEARCH

During this PhD investigation important questions and limitations have been highlighted which are worthy of further research:

- the most important sources of uncertainty highlighted in this dissertation, namely emissions from land use and land use change, shall be further addressed

in order to reduce the overall uncertainty of biofuel chains and improve the reliability of biofuel life-cycle studies outcomes;

- in this dissertation, the GHG assessment of biofuel systems does not account for indirect land use change (iLUC). This aspect is acknowledged as potentially contributing to important environmental impacts, namely GHG emissions, and has recently been the subject of important controversy among the scientific community. Nevertheless, methodological guidelines for iLUC are missing. Moreover, iLUC requires a consequential approach, whose standardization is also still under development (Earles and Halog 2011). An harmonized methodology to account for iLUC issues associated to biofuels is expected at the European level in the near future, enabling a widespread inclusion of iLUC in future biofuel life-cycle studies;
- the assessment of first-generation biofuel systems is not limited to energy and GHG balances, and displacement benefits or drawbacks over petroleum fuels. Other environmental impact categories must also be investigated, especially concerning biofuel effects at local and regional scales. Examples include biodiversity losses associated with energy crop cultivation, eutrophication and acidification from nitrogen and phosphorous fertilizers use, and ecotoxicity and human toxicity due to pesticides application;
- when comparing different biofuel production systems, in particular when incorporating uncertainty, it is advantageous to use a “comparison indicator” that mathematically relates the outputs of the chains to be compared, e.g. through the quotient or difference of the individual contributions to each environmental impact category. A comparison indicator quantifies how significant are the differences between the biofuel systems in each impact category. In this case, it is however essential that shared model structures be respected, otherwise the uncertainty of the difference between biofuel systems can be overestimated (Hong et al. 2010; Plevin 2010). This means that the shared sets of parameters – e.g. global warming potentials for GHGs, production of electricity and fossil fuels – must be computed simultaneously for the biofuel systems under evaluation;

- the introduction of allocation coefficients has a damping effect on the computed parameter uncertainty ranges of outputs, which results in artificially narrow uncertainty ranges for output values. On the other hand, when the substitution method is used, not only the overall uncertainty of the biofuel system is taken into account in the output values but also the uncertainty associated to the products displaced by biofuel co-products. This issue deserves further clarification when both allocation and substitution approaches are used in biofuel life-cycle assessments incorporating uncertainty;
- in this dissertation, the contribution of agricultural machinery to the assessment of capital goods has not been included based on potential subcontracting practices that reduce the impact of farming equipment. Further study with detailed information on this issue is needed to confirm this assumption.

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Appendix

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APPENDIX

A. FEEDSTOCK AND ENERGY DATA

Table A.1. Parameter values for fertilizers, pesticides and seeds production.

Parameter				Source
Energy use and emissions				
N fertilizer production	*MJ kg^{-1} N*	*kg CO_2eq kg^{-1} N*		
	51.9	3.0		*Poitrat et al. (1998)*
	51.7	8.5		*LCA Food DK (1999)*
	79.8	11.6		*Patyk & Reinhardt (2000)*
	40.6	6.7		*Punter et al. (2004)*
	40.6 ± 6.7	6.34 ± 0.3		*Mortimer & Elsayed (2006)*
	41.1 – 60.3	2.6 – 8.5		*Nemecek et al. (2007)*
		3.5 – 5.4		*Elsgaard (2010)*
P_2O_5 fertilizer production	*MJ kg^{-1} P_2O_5*	*kg CO_2eq kg^{-1} P_2O_5*		
	19.5	1.33		*Poitrat et al. (1998)*
	19.8	1.18		*LCA Food DK (1999)*
	18.0	1.18		*Patyk & Reinhardt (2000)*
	19.0	1.35		*Kim & Dale (2004)*
	8.9	0.40		*Punter et al. (2004)*
	15.8	0.71		*Mortimer & Elsayed (2006)*
	11.2 – 45.7	0.76 – 2.61		*Nemecek et al. (2007)*
K_2O fertilizer production	*MJ kg^{-1} K_2O*	*kg CO_2eq kg^{-1} K_2O*		
	9.1	0.59		*Poitrat et al. (1998)*
	11.5	0.67		*LCA Food DK (1999)*
	11.3	0.68		*Patyk & Reinhardt (2000)*
	9.0	0.65		*Kim & Dale (2004)*
	7.8	0.38		*Punter et al. (2004)*
	9.3	0.46		*Mortimer & Elsayed (2006)*
	8.0	0.49		*Nemecek et al. (2007)*
Pesticides production	*MJ kg^{-1}*	*kg CO_2eq kg^{-1}*		
	77.5	2.54		*Poitrat et al. (1998)*
	285.9	8.17		*Patyk & Reinhardt (2000)*
	439.7	24.73		*Kim & Dale (2004)*
	274.1	5.38		*Mortimer & Elsayed (2006)*
	199.4	7.35		*Nemecek et al. (2007)*
Seeds prod. (rapeseed)	*MJ kg^{-1}*	*kg CO_2eq kg^{-1}*		
	7.8	0.61		*Elsayed et al. (2003)*
Seeds prod. (wheat)	*MJ kg^{-1}*	*kg CO_2eq kg^{-1}*		
	5.0	-		*Richards (2000)*
	13.5	0.85		*Elsayed et al. (2003)*
Seeds prod. (sugar beet)	*MJ kg^{-1}*	*kg CO_2eq kg^{-1}*		
	35.5	1.8		*Elsayed et al. (2003)*

Table A.2. Selected probability distributions for fertilizers and pesticides production.

Parameter	distribution	mean	std dev
Energy use in fertilizer production [MJ kg^{-1}]			
N fertilizer	Normal	55	5.5
P_2O_5 fertilizer	Normal	18	5
K_2O fertilizer	Normal	10	0.6
Emissions in fertilizer production [kg CO_2eq kg^{-1}]			
N fertilizer	Normal	3.7	2.2
P_2O_5 fertilizer	Normal	1.1	0.3
K_2O fertilizer	Normal	0.6	0.1
Energy use in pesticides production [MJ kg^{-1}]	Normal	270	81
Emissions in pesticides production [kg CO_2eq kg^{-1}]	Normal	6.8	2.7

Table A.3. Feedstock data (Elsayed et al. 2003; Mortimer et al. 2006).

Parameter	*MJ kg^{-1}*	*kg CO_2eq kg^{-1}*
Hexane	52.1	0.56
Phosphoric Acid	11.4	0.80
Sodium Hydroxide	19.9	1.20
Methanol	38.1	2.80
Sulphuric acid	2.4	0.14
Alkaline catalyst	43.3	2.44

Table A.4. Energy data used.

Parameter	*MJ_{prim} MJ^{-1}*	*kg CO_2eq MJ^{-1}*
Natural Gas	1.1	0057
Electricity	2.7	0.14

B. dLUC MODELING CAPTURING VARIABILITY OF AGRICULTURAL PRACTICES

GHG savings: Rapeseed oil *vs.* Fossil diesel

Life-cycle GHG emission savings of rapeseed oil with respect to fossil diesel are shown in Fig. 5.5. Rapeseed oil from crops cultivated on previous grassland results in higher emissions than the other two scenarios; moreover, “grassland to cropland” emissions most certainly cannot meet the 35% target of EPC (2009), except for the mass allocation approach. It must be remind however that despite being a straightforward method, mass allocation is very often a meaningless approach, namely when energy systems or market principles come into play. The more realistic substitution method shows that the use of grassland for rapeseed cultivation results, on average, in higher emissions than the fossil diesel reference system (negative GHG emission savings in Fig. 5.5). In contrast, switching from cropland or set-aside land to rapeseed cultivation results in emission savings which are most probably above the directive’s target, particularly in the first case. Nevertheless, a very high parameter uncertainty as shown in Fig. 5.5 leads to wide ranges of GHG savings; it also overrides differences between co-product approaches.

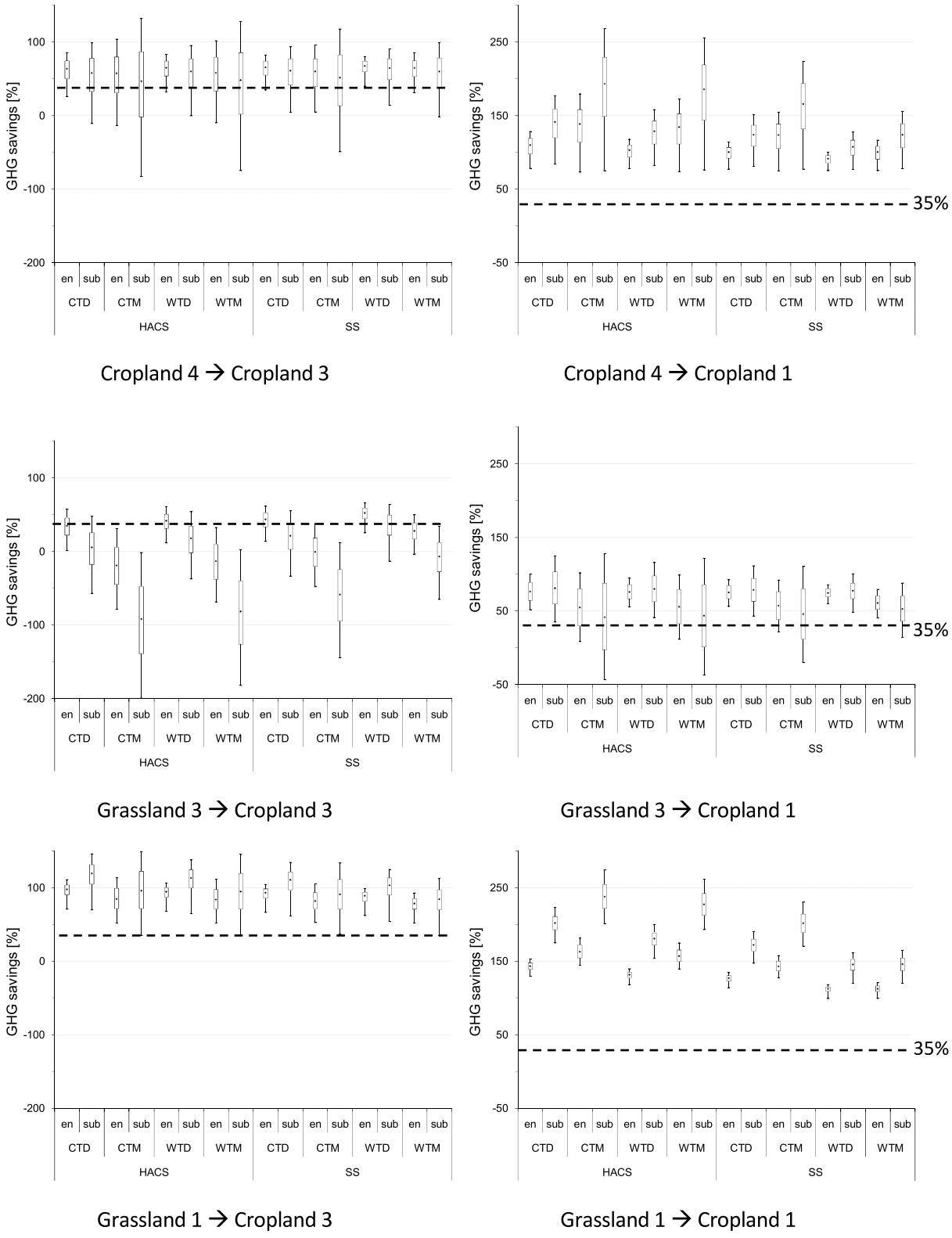

Fig. B.1. GHG emission savings (RO vs. Fossil Diesel). The dashed line indicates the 35% minimum level of biofuel GHG savings for the purposes of EPC (2009). Negative savings indicate net GHG emissions. (en: energy allocation; sub: substitution method).